Bridge River Project Water Use Plan

Lower Bridge River Aquatic Monitoring

Implementation Year 7

Reference: BRGMON-1

BRGMON-1 Lower Bridge River Aquatic Monitoring, Year 7 (2018) Results

Study Period: April 1 2018 to March 31 2019

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July 25, 2019

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Executive Summary

A third year of high flow monitoring was conducted in 2018. The peak flow release from Terzaghi Dam was 102 $m^3 \cdot s^{-1}$ and average flows for the year were 18 $m^3 \cdot s^{-1}$. The 2018 high flow period began in the second week of May, peaked at the end of June, and was ramped back down across the month of July, reaching Trial 2 hydrograph flows on 1 August (high flow duration = 83 days). Outside of the high flow period, the flow releases conformed to the Trial 2 hydrograph from the Lower Bridge River (LBR) flow experiment.

As reported last year, 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, 2017 and 2018 was intended to characterize some of these effects in reaches 2, 3 and 4 in the first three years 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 2018 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 and 2017) years. Three core monitoring activities were continued in 2018: 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 (including 2 new index sites established in the Yalakom River); and 3) a fall standing stock assessment to estimate the relative abundance and distribution of juvenile salmonids in the study area. A fourth activity (i.e., periodic sampling to monitor juvenile salmonid growth), which had been included in previous study years, was not completed in 2018 as per a recommendation in the Year 6 (2017) report that was accepted by St'at'imc Eco-Resources (SER) and BC Hydro (Sneep et al. 2018).

Some additional monitoring components to assess other impacts of the modified operations at Terzaghi Dam (i.e., high flows) were also conducted in 2018. These activities included: water quality monitoring, kokanee entrainment surveys, bank erosion/substrate recruitment assessments, juvenile salmonid habitat use and displacement surveys, and high flow ramp down monitoring and stranding risk assessment.

On balance, the net effects of the high flows released from 2016 to 2018 have been 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). In general, the 2018 results were very consistent with the data for the other high flow years (2016, 2017). 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 water column depth (up to 1.42 m at km 36.8 above the Trial 2 peak) and mid-channel velocities (unmeasured), which reduced the amount of suitable rearing habitat per wetted area;
- 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. Also, Trial 3 temperatures tended to be warmer on average in the months of January and February than the previous flow trials, which may have accelerated incubation conditions for coho by up to a month in Reach 4, where high use for spawning by this species has been observed under BRGMON-3 monitoring;
- Overall benthic invertebrate density has declined by an average of 73% following the high flows in 2016, 2017 and 2018 (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 76% compared to the Trial 2 average (reductions by species-age class were: -76% for steelhead fry, -71% for steelhead parr, equivalently low abundance for chinook fry, and -89% for coho fry) – Notably, coho fry were less abundant in the study area during the high flow years than chinook fry;
- Juvenile salmonid biomass trends mirrored the trends in abundance since differences in mean size for each species and age class were generally less significant among the trials than the changes in mean abundance;
- The addition of the 2018 results did not substantively change the high flow (Trial 3) abundance or biomass estimates for any species and age class. The differences in these metrics among the high flow years were very small relative to the differences among trials.
- Stock-recruitment curves for Trial 3 (high flows) suggest 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 data at low escapements are required to inform the initial slope of the curves and reduce uncertainty about the number of spawners required to fully seed the river;
- 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 (# of fish per 1 $m^3 \cdot s^{-1}$ flow change) appears to be lower at flows above ~13 $m^3 \cdot s^{-1}$ than below.

- Entrainment of kokanee from Carpenter Reservoir into the Lower Bridge River channel occurred in both 2016 (*n*=83 observed) and 2017 (*n*=48 observed). Entrained kokanee were not observed during high flow monitoring in 2018.

Results that noted positive effects or changes associated with the high flows included:

- Seasonal enlargement of the wetted area of the channel, which allowed additional recruitment of new substrate materials from the toe of alluvial slides adjacent to the channel in reaches 2, 3 and 4;
- 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 at higher flows. Cell counts increased over the Trial time series in all reaches. In Reach 2 mean cell counts changed from 7914 cells x $10^6 \cdot m^{-2}$ in Trial 0 to 55,836 cells x $10^6 \cdot m^{-2}$ in Trial 3. The same pattern of about a seven-fold increase in cell density occurred in Reach 3 between Trials 0 and 3 and in Reach 4 between Trials 1 and 3. The increased algae production under Trial 3 flows may have resulted from warmer water temperatures observed across the reaches during summer (preceding sampler deployment) relative to the other trials, and/or the dramatic reduction in aquatic invertebrates, some of which graze on the algae;
- Higher mean size of juvenile steelhead, chinook and coho in each reach (though there was significant overlap in standard deviations in some cases). This result was potentially caused by a few different factors, such as: 1) substantially reduced abundance since food sources (invertebrates) were also substantially reduced (see above); 2) the warmer temperatures during the summer rearing period which likely facilitated growth; and 3) the high flows likely selected for larger fish since they are more mobile and capable of competing for habitat space, while smaller fish may be more readily displaced downstream.
- Evidence of significant use of the enhanced off-channel habitats at Bluenose (in Reach 4) and Applesprings (in Reach 1), particularly by coho fry and mykiss fry and parr when they were sampled following the high flow period in 2018. Densities (i.e., fish/100 m^2) in these habitats were on par with mainstem densities in these reaches during the Trial 1 and Trial 2 flow years when fish abundance estimates were much higher overall, suggesting that these sites serve as refuge habitats that were largely sheltered from the effects of the high flows in the mainstem.

Summary of BRGMON-1 Management Questions and Interim (Year 7 – 2018) Status

Primary Objectives	Management Questions	Year 7 (2018) Results To-Date
Core Components: To reduce uncertainty about the relationship between the magnitude of flow release from the dam and the relative productivity of the Lower Bridge River aquatic and riparian ecosystem. To provide comprehensive documentation of the response of key physical and biological indicators to alternative flow regimes to better inform decision on the long term flow regime for the Lower Bridge River. The scope of this program is limited to monitoring the changes in key physical, chemical, and biological productivity indicators in reaches 2, 3, and 4 of the Lower Bridge River aquatic ecosystem.	How does the instream flow regime alter the physical conditions in aquatic and riparian habitats of the Lower Bridge River ecosystem?	• The biggest gains in wetted area were achieved by the wetting of Reach 4 and the augmentation of flows in Reach 3 by the Trial 1 and 2 treatments. 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 $m^3 \cdot s^{-1}$ (Ellis et al. 2018). High flows also recruited material from edge sources (e.g., the toe of alluvial slopes). • Water temperatures under all trial flows were cooler in the spring and warmer in fall relative to the Pre-flow (Trial 0) profile. Under high flows in 2016, 2017 and 2018 (Trial 3) 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). Trial 3 temperatures also tended to be warmer on average in the months of January and February, which may have accelerated incubation conditions for coho by up to a month in Reach 4.
	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 3 and lowest in Reach 4. • Flows during Trials 1 and 2 produced what might be called optimum conditions for the benthic communities. The average 73% decline in invertebrate density and low diversity associated with Trial 3 flows showed that physical conditions associated with high peak flow, potentially including scour and bed movement, did not favour the benthic communities. Given that benthos found in the Lower Bridge River includes common fish food organisms, the Trial 3 flows caused a decline in the food available to fish at the time of measurement in the fall months. • 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.
	How do changes in physical conditions and trophic productivity resulting from flow changes together influence the recruitment of fish populations in Lower	• Juvenile salmonid abundance was highest (overall) under the Trial 1 and 2 flow regimes (in general, production between them was near equivalent, but both impacted chinook recruitment). Relative to the previous flow treatment, the high flows in 2016, 2017 and 2018 reduced salmonid abundance by 76%. Reductions for steelhead and coho fry were by 76% and 89%, respectively. Steelhead parr abundance was 71% lower and 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 possible that habitats were fully seeded in most study years; however, more data

Primary Objectives	Management Questions	Year 7 (2018) Results To-Date
	Bridge River?	at low escapements 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 for some standard deviations). Lower fish abundance likely resulted in reduced competition for the available food resources.
	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 2018 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 – 2018 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 Monitoring	Do flow releases from Terzaghi Dam under the modified flow regime affect water quality or cause erosion in the Lower Bridge River? If so, what are the potential effects on fish and what mitigation options are available?	• The high flows in 2018 resulted in elevated water temperatures, turbidity and %TGP levels (relative to background) in reaches 3 and 4, and caused some erosion and substrate recruitment along the wetted edge at various sites that were monitored in reaches 2 – 4. • Temperature changes were within optimal ranges for juvenile salmonid rearing cited in the literature, the turbidity changes in 2018 were lower than 2017, the erosion was at existing alluvial slides, and the %TDG levels were below the thresholds for triggering a mitigation response according to Table 1 of BC Hydro's Total Dissolved Gas Management Strategy (i.e., BC Hydro 2014). • As such, within the scope of information available in 2018, we don't have any data to suggest a direct adverse effect of the measured temperatures, turbidity levels, %TDG saturation or bank erosion across the high flow period on fish in 2018.
High Flow Ramp Down Monitoring and Stranding Risk Assessment	How does the risk of fish stranding during LBR ramp downs vary with discharge?	• Above a threshold of ~13 $m^3 \cdot s^{-1}$, the fish stranding risk (per 1 $m^3 \cdot s^{-1}$ increment of flow change) was consistently low (or occasionally moderate). Conversely, below the 13 $m^3 \cdot s^{-1}$ threshold, the fish stranding risk was more consistently high. • This difference likely provides the opportunity to continue to implement (and monitor) faster ramp rates for higher flows (>13 $m^3 \cdot s^{-1}$) * Important caveat: juvenile fish abundance was substantially reduced overall in 2016 – 2018, which likely affected salvage results following high flows during those years.
	How does the risk of fish stranding during LBR ramp downs vary by reach?	• Under previous flow trials (≤15 $m^3 \cdot s^{-1}$), differences in the number of fish salvaged (per 100 m^2) among reaches was significant. Reach 4 densities were more than double Reach 3 densities. • Differences among reaches in the high flow range (>15 $m^3 \cdot s^{-1}$) were also apparent but they were smaller. Slightly higher densities were observed in Reach 3, followed by Reach 4, and then Reach 2 and Reach 1.
	How does the risk of fish stranding during	• At higher ramp rates up to 4.1 cm/hr implemented in 2017 and 2018, there was no appreciable difference in fish stranding risk relative to lower rates (≤2.5 cm/hr) across the high flow range tested: 102.0 to

Primary Objectives	Management Questions	Year 7 (2018) Results To-Date
	LBR ramp downs vary with ramping rate and stage change?	44.7 $m^3 \cdot s^{-1}$). • These results, while still 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 abundances of juvenile salmonids in 2017 and 2018 were low overall, which could have influenced results.
	How does the risk of fish stranding during LBR ramp downs vary by river bank?	• At high flows, site distribution was equal (50% river left; 50% 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.
	Are there opportunities to minimize or mitigate the risk of fish stranding during ramp downs in the Lower Bridge River?	• The primary opportunity (or most conservative approach) for minimizing or mitigating the risk of fish stranding is by implementing the ramping rates referenced in the WUP (i.e., ≤2.5 cm/hr) and having fish salvage crews actively salvaging fish in each of the reaches downstream of the dam. • In some cases, such as in the past 3 high flow years, there can be additional rationale for ramping the flows down faster in order to reach more optimal summer rearing flows (i.e., ≤15 $m^3 \cdot s^{-1}$) more quickly following peak flows. With the data for high flows available from 2016 to 2018, there is some evidence for when faster ramping rates can be applied without unduly increasing fish stranding risk. • Implementation of faster ramp rates should be accompanied by ramp monitoring and fish salvaging to improve certainty about effects on stranding risk.
Juvenile Salmonid Habitat Availability and Displacement	How does juvenile salmonid habitat availability in the Lower Bridge River change with discharge under the modified flow regime?	• Based on the results of the pilot-level sampling in 2018, there is insufficient information at this point to answer this management question (i.e., from the field data alone). • Depths and velocities were highly variable across the different mainstem sites and tended to be more consistent in the off-channel sites • There were some sites (both high quality and low quality) that provided suitable depths and velocities for rearing (based on LBR HSI curves for coho, chinook and mykiss) across the full range of high flows. • A better approach to addressing this question may be to use BC Hydro's Telemac2D model
	How does habitat use by juvenile salmonids change with discharge under the modified flow regime?	• For the fry age class of coho and chinook, the densities in the pre-selected mainstem sites appeared to diminish at flows between 28 and 70 $m^3 \cdot s^{-1}$ on the ascending limb of the hydrograph in 2018. • Mykiss fry densities were low across the ascending limb and peak of the high flow hydrograph, and then increased in the latter part of the descending limb (i.e., between 82 and 27 $m^3 \cdot s^{-1}$), which was likely related to emergence timing for this species. • Mykiss parr densities tended to be highest on the ascending limb of the hydrograph (i.e., up to 70 $m^3 \cdot s^{-1}$), and they appeared more tolerant of high flows than the coho and chinook fry. • There were no substantial changes in density in the off-channel sites that would point to significant immigration or emigration during the high flow period in 2018, particularly for fry.

Primary Objectives	Management Questions	Year 7 (2018) Results To-Date
Substrate Mobilization, Deposition and Composition Monitoring	To what extent does substrate movement under the modified flow regime affect the distribution, availability or suitability of juvenile rearing in the Lower Bridge River and what potential mitigation opportunities exist to minimize or mitigate any measured effects?	• There were no data available within the scope of the BRGMON-1 program in 2018 for answering this question. Refer to the Year 6 report (Sneep et al. 2018) and the latest report by KWL on their substrate mobilization, deposition and composition monitoring (Ellis et al. 2018) for some relevant information pertaining to this question. • We understand that work is also being conducted by BC Hydro using the Telemac2D model to make predictions about the amount and distribution of rearing habitat area for the various salmonid species across the range of 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 two to three times per decade (Figure 1.1). The flooding and subsequent dewatering associated with these events inevitably had impacts on the LBR ecosystem.

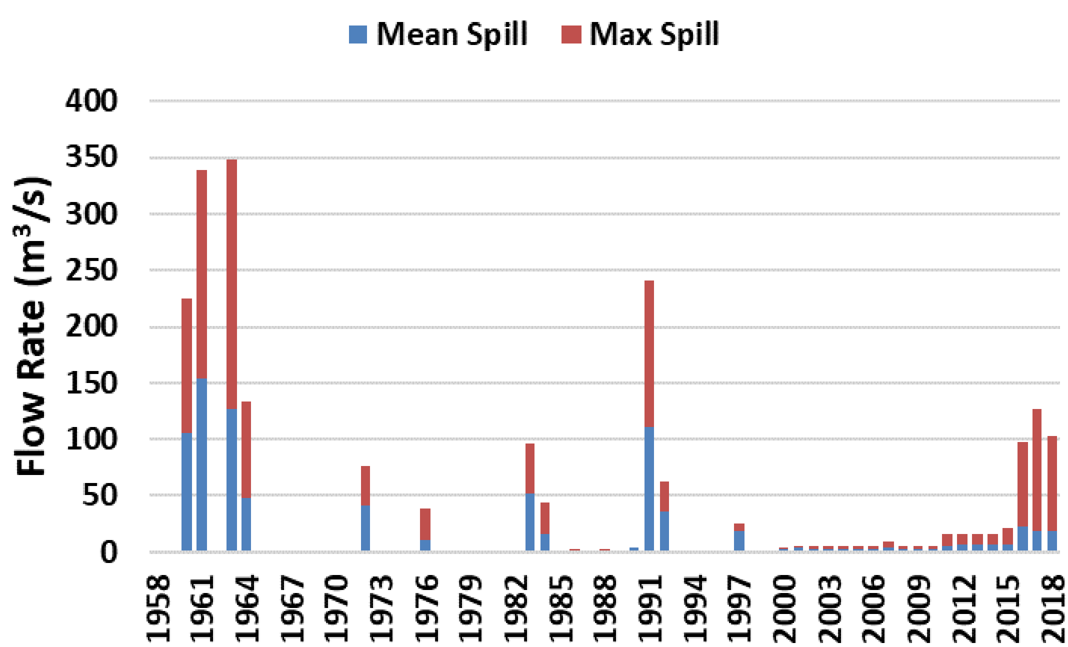


Figure 1.1 Frequency of spill and flow release events from Terzaghi Dam into the Lower Bridge River following impoundment in 1960.

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 (Sneep and Hall 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.

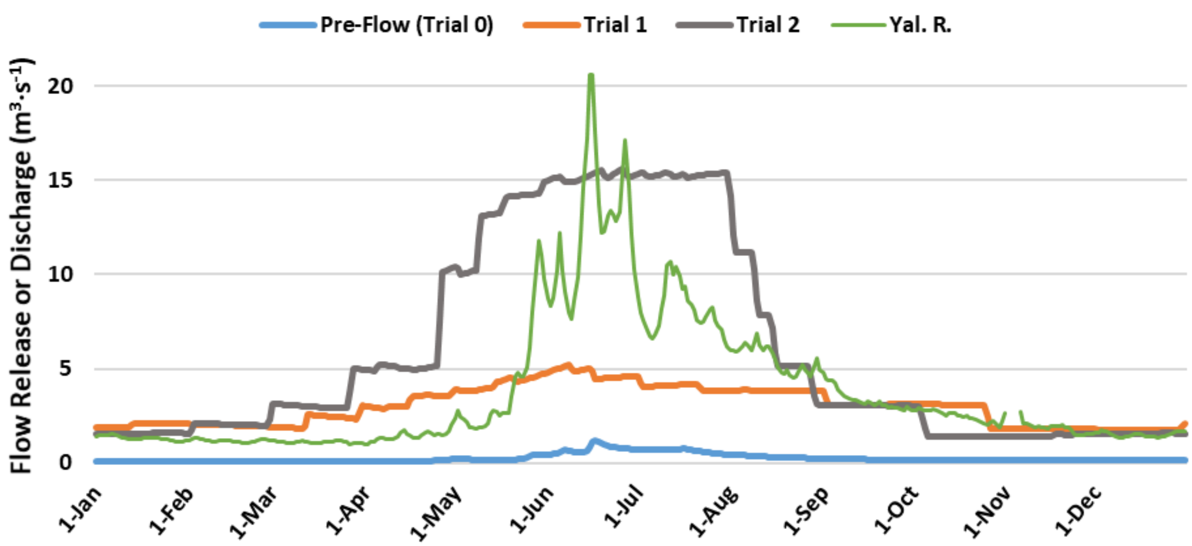


Figure 1.2 Mean daily releases from Terzaghi Dam for Trial 1 and Trial 2 during the flow experiment. Typical hydrograph shapes during the Pre-flow period and for the unregulated Yalakom River discharges are included for reference.

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 and 2018. Mean annual flows from the dam were approximately 22, 19 and 18 $m^3 \cdot s^{-1}$ (peak flows = 97, 127 and 102 $m^3 \cdot s^{-1}$) in 2016, 2017 and 2018, respectively (Figure 1.3). These high flow years are collectively referred to as "Trial 3" in the context of the analyses within this report.

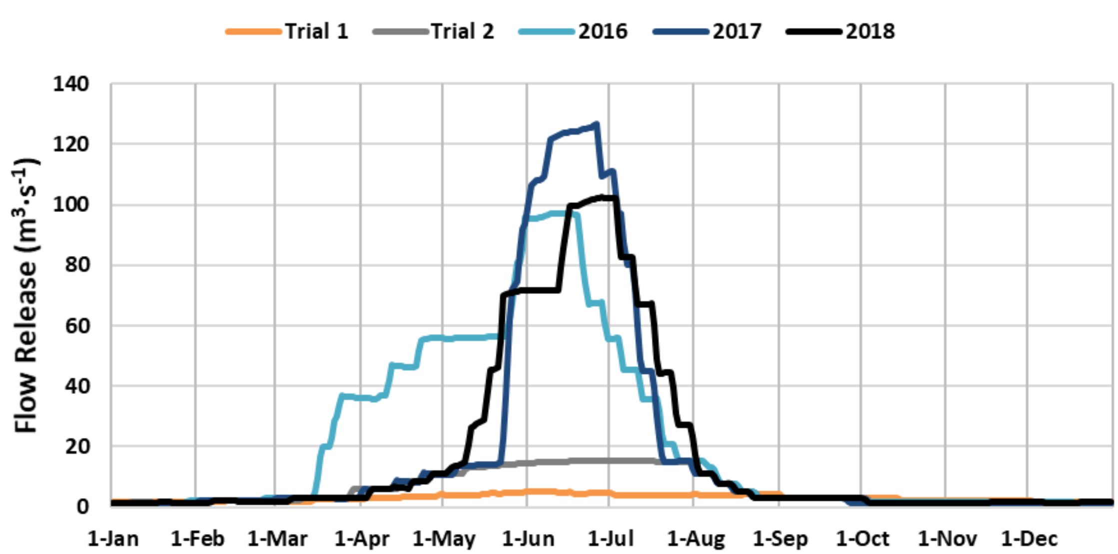


Figure 1.3 Hydrograph shapes for the high flows released from Terzaghi Dam into the Lower Bridge River channel in 2016, 2017 and 2018. Mean daily releases for the Trial 1 and Trial 2 hydrographs are shown for context.

Peak flows from 2016 to 2018 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 on 17 March, peaked in mid June, and returned to Trial 2 levels by 25 July (2016 high flow duration = 131 days). The high flows in 2017 had a higher peak, but a shorter duration relative to 2016: Flows increased above the Trial 2 hydrograph on 24 May, peaked across the month of June, and were ramped back down to Trial 2 levels on 21 July (2017 high flow duration = 59 days). High flows in 2018 began on 10 May, peaked in late June, and were ramped back down to Trial 2 levels on 1 August (2018 high flow duration = 83 days). Outside of the high flow period, the flow release from mid summer through fall and winter has been identical to the Trial 2 hydrograph shape during each high flow year (to-date).

Figure 1.4 shows mean trial flows on a logarithmic scale to compare differences in the shapes of the flow release hydrograph between trials. Trial 3 produced a pronounced bell-shaped hydrograph with steep ascending and descending limbs and highest peak release among trials. Trial 1 shape was a flattened bell-shaped hydrograph appearing more like a shallow dome with low slopes on the ascending and descending limbs and lowest peak release among trials. The Trial 2 hydrograph was in between, having a moderate bell shape, moderate rates of ascending and descending limbs and peak water releases in between Trials 1 and 3.

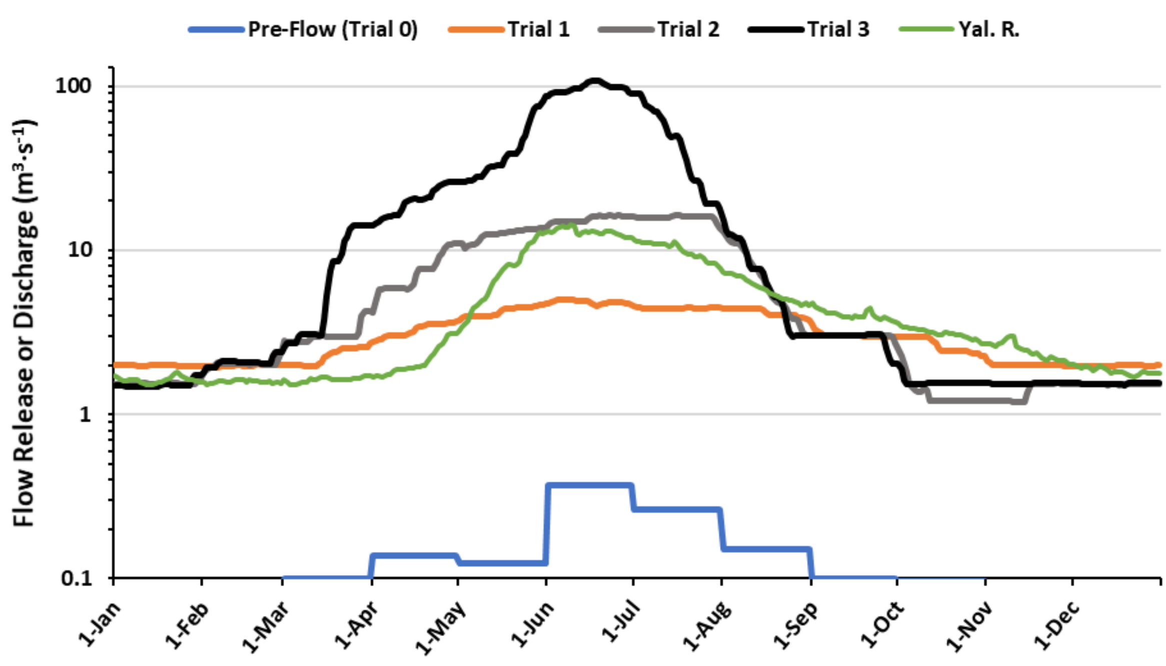


Figure 1.4 **Mean daily flow release from the Terzaghi Dam among all years in each flow Trial. Mean daily flow among all years (1996 – 2018) in the Yalakom River is shown for reference. Note the log scale on the Y axis.**

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

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

River	Trial number	Flow statistic ± standard deviation		
		Mean annual water release or flow ($m^3 \cdot s^{-1}$)	Average minimum water release or flow ($m^3 \cdot s^{-1}$)	Average peak water release or flow ($m^3 \cdot s^{-1}$)
Lower Bridge	0 (1996 – 1999)[a] (n=4)	0.6 ± 1.3	0	6.3 ± 12.5
Lower Bridge	1 (2001 – 2010)[a] (n=10)	3.1 ± 0.2	1.8 ± 0.07	5.4 ± 1.1
Lower Bridge	2 (2012 – 2015)[a] (n=4)	6.1 ± 0.3	1.1 ± 0.7	16.8 ± 2.6
Lower Bridge	3 (2016 – 2018) (n=3)	19.5 ± 2.1	1.4 ± 0.05	109 ± 15.7
Yalakom	0 (1996 – 2000) (n=4)	5.1 ± 1.2	1.2 ± 0.13	25.1 ± 12.7
Yalakom	1 (2001 – 2011) (n=10)	4.2 ± 0.9	1.2 ± 0.19	22.5 ± 10.7
Yalakom	2 (2012 – 2015) (n=4)	4.5 ± 0.5	0.7 ± 0.4	21.9 ± 5.1
Yalakom	3 (2016 – 2018) (n=3)	5.2 ± 0.4	0.9 ± 0.3	25.1 ± 1.8

[a] Years 2000 and 2011 are omitted because they were incomplete years for calculations of flow statistics.

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 magnitude from Terzaghi Dam. The management hypotheses are:

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

1.4.3. Modified Operations (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.

Water Quality, Erosion and Entrainment Monitoring

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

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

Fish Salvage and Stranding Risk Assessment

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

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

Juvenile Salmonid Habitat Availability and Displacement

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

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

Substrate Mobilization, Deposition and Composition Monitoring

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

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

1.5. Study Area

The Bridge River drains a large glaciated region of the Coast Range of British Columbia and flows eastward, 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 1.2. 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). Starting in 2018, monitoring of aquatic productivity metrics (periphyton and benthos) was extended to include sites in the lower portion of the Yalakom River to allow comparison between the flow controlled Lower Bridge River and the unregulated Yalakom River. The overall study area is illustrated in Figure 1.5.

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

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

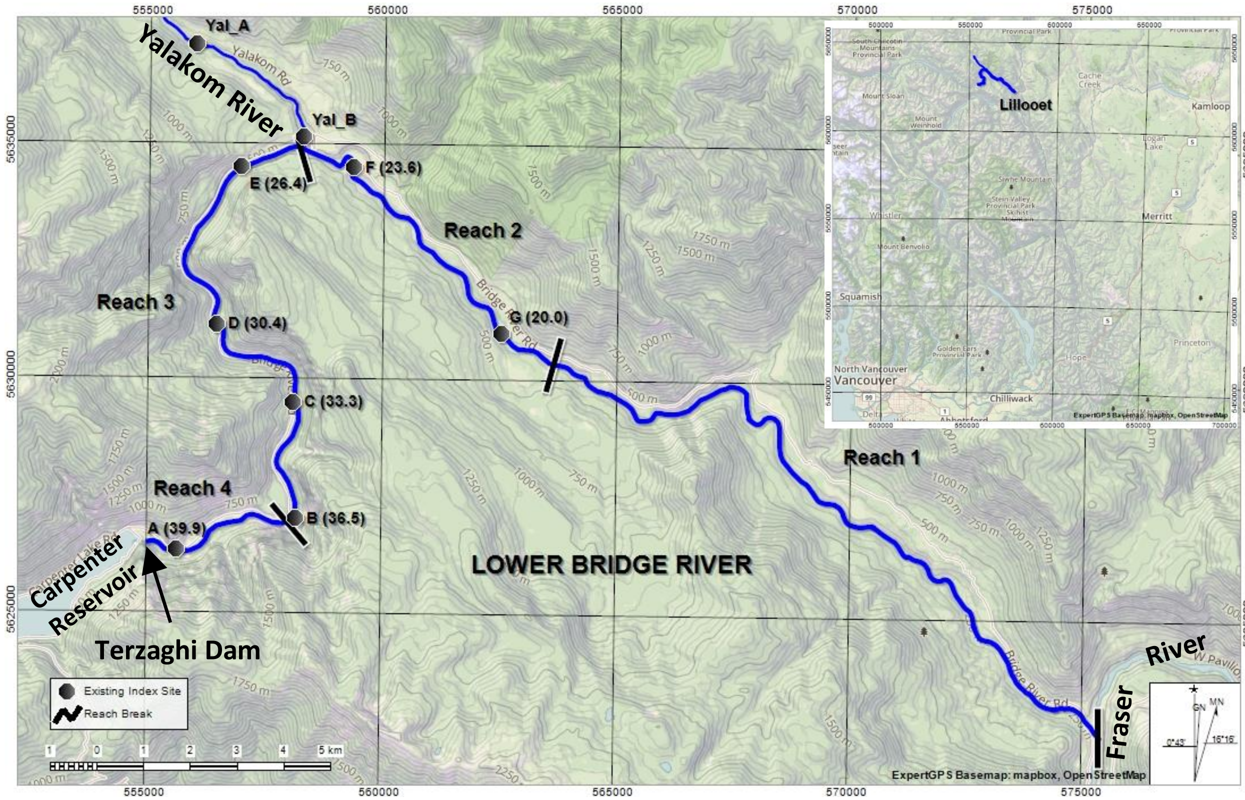


Figure 1.5 The Lower Bridge River downstream of Terzaghi Dam near Lillooet, British Columbia. Reaches are labelled 4 (upstream) through 1 (downstream). Index sampling sites are labelled as distances upstream of the Fraser River and correspond to the following letters in some of the figures below: 39.9 km (A), 36.5 km (B), 33.3 km (C), 30.4 km (D), 26.4 km (E), 23.6 km (F) and 20.0 km (G). Yal_A and Yal_B are two new index sites established in the Yalakom River in 2018. The inset map in the top-right corner frames the location of the sampling area within the context of southwestern British Columbia.

Prior to initiation of the continuous flow release at the start of the flow experiment (i.e., August 2000), Reach 4 was the previously dry section immediately below the dam (length = 3.2 km). Tributary inflows to this reach 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 2018 was conducted between April and December according to each monitoring component (Table 1.3). Certain components that were measured by loggers (i.e., water temperature, river stage, and discharge from the dam) were recorded year-round. This report focusses on the data collected in 2018; however, comparisons or context from previous years and flow trials are included where relevant and available.

Table 1.3 Summary of data to be included in BRGMON-1 analysis and reporting for monitoring year 2018. Components that have prior years of data are noted.

Task	Components	2018 Period	Prior Years of Data[1]
Physical Parameter Monitoring	Water temperature; river stage; discharge	Year-round	1996 to 2017
Water Chemistry	Nutrients; alkalinity; pH	4 Oct & 4 Dec	1996 to 2017
Primary & Secondary Productivity	Periphyton accrual; benthic invertebrate diversity & abundance	25 Sep to 15 Nov (LBR)	1996 to 2017
		27 Sep to 21 Nov (Yalakom)	NA
Juvenile Salmonid Abundance	Annual standing stock assessment	4 to 15 Sep	1996 to 2017
	Juvenile fish density in off-channel habitats	15 & 28 Sep	NA
WUP Ramp Down Monitoring	Stage monitoring; fish salvage	2-22 Aug & 2-3 Oct	2011 to 2017
High Flow Monitoring	Kokanee entrainment; water quality sampling; sediment erosion & deposition; fish stranding site reconnaissance	6 May to 1 Aug	2016, 2017
Juvenile Salmonid Habitat Availability & Displacement	Single-pass, open site electrofishing at pre-selected low- and high quality rearing sites	18 dates between 9 May to 30 Jul	NA
High Flow Ramp Down & Stranding Risk Assessment	Stage monitoring; fish salvage at flows >15 m^3/s	4 Jul to 1 Aug	2016, 2017

[1] Results of analyses for prior years of monitoring will only be included in this annual report where relevant for providing context to the 2018 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 control site was not originally included, the study design has relied 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 2018 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, 2017) periods. Three 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; and 3) a fall standing stock assessment to estimate the relative abundance and distribution of juvenile salmonids in the study area. A fourth activity (i.e., periodic sampling to monitor juvenile salmonid growth), which had been included in previous study years, was not completed in 2018 as per a recommendation in the Year 6 (2017) report that was accepted by St'at'imc Eco-Resources (SER) and BC Hydro (Sneep et al. 2018). Instead, assessment of fish densities in two off-channel habitats during the stock assessment period was added in 2018.

Activities 1) and 2) were conducted at the usual seven index sites in the LBR 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 (see Figure 1.5). In addition, two new sites were added in the Yalakom River (Yal_A and Yal_B) in 2018, at 3.6 km and 0.2 km upstream of the LBR-Yalakom River confluence, respectively. Inclusion of these new sites was intended for documenting periphyton accrual and benthic invertebrate diversity and abundance in this important tributary and invertebrate recruitment source for the lower reaches of the LBR. The fall standing stock assessment was conducted at 36 sites (during the Pre-flow period) and 49-50 sites (during the 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.

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

Trial	Years	Reaches	Seasons when samples were collected	Target mean annual flow release from Terzaghi Dam ($m^3 \cdot s^{-1}$ ± 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	0.5 ± 1.1
Trial 1	August 2000 – 2005	2, 3, 4	Spring Summer Fall	3 ± 5%	3.0 ± 0.3
	2006 – 2010	2, 3, 4	Fall		
Trial 2	2011 – 2015	2, 3, 4	Fall	6 ± 5%	6.2 ± 0.4
Trial 3	2016 – 2018	2, 3, 4	Fall	No target [a]	19.5 ± 2.1
	2018	Yalakom		n/a	n/a

[a] Trial 3 flows were a variance from Trial 2 resulting from reduction of water storage in Downton Reservoir and issues limiting diversion of flow above Terzaghi Dam to the generating stations at Shalalth. Flow excursions above the Trial 2 hydrograph (in terms of magnitude and duration) depend on snowpack and inflows during each Trial 3 year.

Field data collection for the Lower Bridge River Aquatic Monitoring Program (BRGMON-1) and the additional high flow monitoring components in 2018 were conducted by members of Coldstream Ecology Ltd. and Xwísten. The field studies project manager Alyson McHugh and members of her team also managed the collection of data, reporting and analysis for most of the Trial 2 years (i.e., 2012 to 2015), and the first high flow (Trial 3) year in 2016 (McHugh and Soverel 2013, 2014, 2015, 2017; McHugh et al. 2017).

2.1.1. Discharge, River Stage and Water Temperatures

Discharge rates were either provided or estimated according to location in the study area. Flows in Reach 4 (after initiation of the flow release) were comprised entirely of dam discharge since tributary inputs to this reach are very minor and ephemeral. As such the discharge data for this reach were based on the flow release values alone, which were provided by BC Hydro Power Records (as hourly values). Flows at each index site in reaches 3 and 2 were estimated using a plug-flow approach (described by equations 1 and 2, below) based on tributary drainage area coupled with known Yalakom River discharge data provided by Water Survey of Canada

(Gauge 08ME025). Mean daily, site-specific discharge estimates were calculated for each index site according to the following formulas:

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

where

$Q_{trib,\,d}$ = discharge of Lower Bridge River tributary on day (d) – see list of tributaries provided in Section 1.5;

$Q_{yal,\,d}$ = discharge of the Yalakom River on day (d) -- data provided by Water Survey of Canada;

A_{trib} = drainage area of Lower Bridge River tributary (estimated from a 1:50,000 topographic map); and,

A_{yal} = drainage area of the Yalakom River above the WSC gauge (estimated from a 1:50,000 topographic map).

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

where

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

$Q_{i-1,\,d}$ = discharge at the next upstream index site (i-1) on day (d); and,

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

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

Site A (km 39.9) = Terzaghi release discharge;
Site B (km 36.5) = Site A discharge + km 36.8 groundwater inflow estimate;
Site C (km 33.3) = Site B discharge + Mission Creek inflow;
Site D (km 30.4) = Site C discharge + Yankee Creek & Russell Springs inflow;
Site E (km 26.4) = Site D discharge + Hell Creek & Michelmoon Creek inflow;
Site F (km 23.6) = Site E + Yalakom River inflow;
Site G (km 20.0) = Site F + Antoine Creek inflow.

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

To evaluate the effects of flow releases on the timing of emergence of chinook and coho salmon fry from spawning gravels we calculated the accumulated thermal units (ATU, as the sum of daily temperatures above 0°C) from the observed average date of peak spawning, using average surface water temperatures for each monitoring station. Median emergence was assumed to occur at 1000 ATU for chinook salmon (Groves et al. 2008) and 500 ATU for coho salmon (Murray et al. 1990; based on development data for 2-5°C water). Peak of spawning was set at September 8 and November 15 for chinook and coho salmon, respectively, based on observations made during streamwalk surveys conducted under BRGMON-3 (Melville et al. 2015).

2.1.2. Periphyton Biomass and Composition

Field Methods

Periphyton was sampled from riffle or run habitats at each LBR index site: A (km 39.9), B (km 36.5), C (km 33.3), D (km 30.4), E (km 26.4), F (km 23.6), G (km 20.0), as well as two sites in the Yalakom River (Yal_A and Yal_B) for the first time in 2018 (see Figure 1.5 for relative locations). 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 – June), summer (August – September), and fall (October – 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-cell Styrofoam (Floracraft Corp., Pomona CA) attached to a plywood plate that was bolted to a concrete block. Styrofoam is a good substratum because its rough texture allows for rapid seeding by algal cells, and the adhered biomass is easily sampled (Perrin et al. 1987). Use of the plates standardized the substrate at all stations and removed variation in biomass accrual due to differences in 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 at all index site locations, during one ca. eight-week accrual period in the fall (i.e., 50 days between 25 Sep and 14 Nov, 2018). The accrual period for the newly established index sites in the Yalakom River was 55 days between 27 Sep and 21 Nov, 2018. 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. The samples were kept frozen from the end of each sampling day until they were analyzed at the lab (ALS Environmental). 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.

A set of depth and velocity measurements were taken for each plate using a top-set wading rod and velocity meter manufactured by Swoffer Instruments, Inc. 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. In 2018, the depths and velocities were measured weekly to better characterize any changes in these parameters across the sampling period.

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). Units were µg chlorophyll-a·cm^{-2}. The highest chlorophyll-a concentration accrued on each plate during the incubation period was considered peak biomass (PB). PB was the metric used to define biomass accrued on a substratum because it is related to cellular growth rate (Bothwell 1989). It was used along with other habitat attributes to find the most important variables contributing to variation in benthic invertebrate assemblages between trials (Section 2.1.4).

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. 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 2.1.2). In 2018, two additional sampling stations were established in the Yalakom in an effort to begin making comparisons among invertebrate metrics between the flow-controlled Lower Bridge River and the unregulated Yalakom River. Each invertebrate sample was collected from 25 – 50 mm size gravel enclosed in a wire basket measuring 30 cm long x 14 cm wide x 14 cm deep (Photo 2.2), with 2 cm openings that was installed in the river for a period of approximately 8 weeks. The basket was similar to that shown by Merritt et al. (2008). The baskets were filled with clean material that was collected from the stream bed or bank and closed using cable ties. To maintain sampling consistency, the same substrates were used in each basket from year-to-year throughout this monitoring program, unless they needed to be supplemented due to spillage or loss during the sampling period. 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 2018, the basket colonization period was 51 days (from 25 Sep to 15 Nov) for LBR index sites A to D, and 52 days (from 25 Sep to 16 Nov) for sites E, F and G. The Yalakom River baskets were installed on 27 Sep and retrieved on 21 Nov, for a colonization period of 55 days.

Water depth and velocity were measured at the upstream end of each benthic invertebrate sampling basket using a top-set wading rod equipped with a velocity sensor manufactured by Swoffer Instruments. In past years, the measurements were generally taken twice: at the start of the sampling series when the samplers were installed, and then again at the end prior to retrieval. In 2018, the depth and velocity measurements were recorded weekly to better characterize any changes in these parameters across the sampling period, the same as was described for the periphyton sampling plates in Section 2.1.2.

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

2.1.4. Habitat Attributes

Attributes that may drive change in patterns of benthic biological assemblages which support fish production were measured or calculated using methods listed in Table 2.2 and further described below.

Table 2.2 List of habitat variables from each site and potentially influencing benthic assemblages in the Lower Bridge River.

Variable name	Units	Description
PB	µg chl-a $\cdot cm^{-2}$	The mean value of highest chlorophyll-a concentration found on replicate periphyton plates during the time of incubation at a given site
Mean water temperature	°C	Mean water temperature recorded every hour during the incubation time series using an Onset Tidbit logger at the site of interest
Mean daily flow release	$m^3 \cdot s^{-1}$	Average of mean daily flow release from Terzaghi Dam during the period of incubation of the plates and baskets
Incubation flow	$m^3 \cdot s^{-1}$	Average of mean daily site-specific flow during the period of sampler incubation (Section 2.1.1)
Disturbance flow	$m^3 \cdot s^{-1}$	Mean site-specific flow during March 1 - August 31 preceding the fall sampling. This metric captures the full spring and summer flow releases between trials including the time between the beginning of ramping up of flow releases and the end of ramping down of flow releases each year.
Peak disturbance flow	$m^3 \cdot s^{-1}$	Peak site-specific flow during March 1 – August 31 preceding the fall sampling. This metric captures the peak spring and summer flow releases between trials.
Mean depth at sampler	cm	Mean water depth at the upstream end of the basket sampler among replicate baskets at a given site
Mean velocity at sampler	$cm \cdot s^{-1}$	Mean water velocity at the upstream end of the basket sampler among replicate baskets at a given site
Habitat type	No units	Mean habitat type which is scored on a scale of 1 to 4 where 1 = Run, 2 = Riffle, 3 = Rapid and 4 = Cascade
Mean habitat wetted area	m^2	Mean wetted area from habitat survey data across all replicates, calculated as the length x the mean width (each described below) of the habitat unit.

Variable name	Units	Description
Mean habitat length	m	Mean length of the habitat unit at each site based on a habitat survey conducted at 1.5 cms in October 2014. For a habitat type where plates and baskets were located (say a riffle), the distance between the upstream and downstream extent of that habitat type was measured. That distance was the habitat length.
Mean wet width	m	Mean wetted width of channel at each site from a habitat survey conducted at 1.5 cms in October 2014. The mean was based on several width measurements between the upstream and downstream extents of the habitat unit.
Mean channel bank-full width	m	This measurement was the width of the channel at top-of-bank. Width was the distance from the eroded edge of one bank to the perpendicular eroded edge of the other bank. Multiple measurements were made within the habitat type where the invertebrate baskets were placed. Mean width was the average value of these measurements within a habitat type at a given site. All data were from past habitat surveys.
Mean channel depth	m	Mean water depth in the middle of the channel from a habitat survey conducted at 1.5 cms in October 2014. It was calculated from multiple measurements in the middle of the channel within the habitat unit where the invertebrate baskets were placed at a given site.
Mean channel velocity	$m\cdot s^{-1}$	Mean water velocity in the middle of the channel from a habitat survey conducted at 1.5 cms in October 2014. It was calculated from multiple measurements in the middle of the channel within the habitat unit where the invertebrate baskets were placed at a given site.
Mean ammonium (NH_4-N) concentration	$\mu g\cdot L^{-1}$	Mean NH_4-N concentration in water samples collected at the start and finish of the incubation period at the site of interest
Mean nitrate (NO_3-N) concentration	$\mu g\cdot L^{-1}$	Mean NO_3-N concentration in water samples collected at the start and finish of the incubation period at the site of interest
Mean DIN concentration	$\mu g\cdot L^{-1}$	Mean DIN (NH_4-N + NO_3-N) concentration in water samples collected at the start and finish of the incubation period at the site of interest

Variable name	Units	Description
Mean SRP concentration	$\mu g \cdot L^{-1}$	Mean soluble reactive phosphorus concentration in water samples collected at the start and finish of the incubation period at the site of interest
Mean TP concentration	$\mu g \cdot L^{-1}$	Mean total phosphorus concentration in water samples collected at the start and finish of the incubation period at the site of interest
Mean TDP concentration	$\mu g \cdot L^{-1}$	Mean total dissolved phosphorus concentration in water samples collected at the start and finish of the incubation period at the site of interest
Mean alkalinity	$mgCaCO3 \cdot L^{-1}$	Mean alkalinity in water samples collected at the start and finish of the incubation period at the site of interest
Mean pH measured in the lab and field	Relative units	Mean pH measured in the lab using a bench top instrument and the field using a field WTW instrument in water samples collected at the start and finish of the incubation period at each site
Mean conductivity	$\mu S \cdot cm^{-1}$	Mean conductivity measured in the field by handheld conductivity instrument in water samples collected at the start and finish of the incubation period at the site of interest
Mean total dissolved solids (TDS) concentration	$mg \cdot L^{-1}$	Mean total dissolved solids concentration in water samples collected at the start and finish of the incubation period at the site of interest
Distance from origin	m	For sites in Reach 4 and 3, distance from origin was distance from the Terzaghi Dam, measured by laser rangefinder. In Reach 2, distance from origin is calculated as: $D_o = D_Y + D_{B34} + D_{B2}$ Where: D_o is distance from origin D_Y is mainstem distance in the Yalakom from the headwaters to the confluence with the Lower Bridge River (measured in Google Earth) D_{B34} is distance in the Lower Bridge River from the dam downstream to the confluence with the Yalakom River (measured by laser rangefinder), and D_{B2} is distance from the confluence of the Yalakom and Lower Bridge River to a given sampling site in Reach 2 (measured by laser rangefinder).

Variable name	Units	Description
Distance from dam	m	Distance from the Terzaghi Dam downstream to a given sampling site, measured by laser rangefinder
Yalakom influence	No units	Binary coding: 1 was assigned if a site was downstream of the confluence with the Yalakom; 0 was assigned if a site was upstream of the confluence with the Yalakom.
Pink run (on or off year)	No units	Binary coding: 1 was an on year, 0 was an off year

Algal PB, defined in Section **Error! Reference source not found.**, was an estimate of autochthonous standing crop food produced for invertebrates that graze on epilithic substrata or collect particles produced by periphyton.

Physical measurements included water temperature and several flow metrics. ***Water temperature*** may modify nymphal growth in aquatic insects (Lillehammer et al. 1989) and contribute to variation in biological assemblages and biomass (Bothwell 1988, Goldman and Carpenter 1974, Scrine et al. 2017, Heggenes et al. 2017). It was measured hourly throughout the plate and basket incubation periods at each sampling site using an Onset Tidbit logger (Bourne, MA).

Several aspects of flow are known to modify stream benthos that are food for fish (e.g. Gore et al. 2001, Nelson and Lieberman 2002, Mérigoux and Dolédec 2004, Dewson et al. 2007, Kennan et al. 2010, and Clausen and Biggs 1997). ***Mean daily flow release from Terzaghi Dam*** and ***site-specific flow during the incubation period***, obtained according to the methods described in Section **Error! Reference source not found.**, showed exposures of the benthic communities to the controlled supply of water to the Lower Bridge River from the Terzaghi Dam and flow-driven habitat conditions during the time of colonization and development of biological assemblages in the samplers. Another hydrologic variable was the average site-specific flow (methods in Section **Error! Reference source not found.**) during the period of March 1 to August 31 before the fall sampling. This metric was called ***disturbance flow*** because it captured the average flows during the bell-shaped flow release period for each trial (Figure 1.4). ***Peak disturbance flow*** was the highest magnitude of flow during each trial.

Water depth and ***water velocity*** at each plate and basket was measured weekly during the periods of incubation using a top-set wading rod and velocity meter (Swoffer Instruments, Inc.). Variables describing channel hydrology and habitat type were also measured during earlier surveys at flows typical of those in the fall during each trial (Higgins and Korman 2000, McHugh and Soverel 2015). They included ***habitat type*** (riffle, pool, run, etc.) and ***mean habitat wetted area*** defined by ***habitat length*** and ***wetted width.*** Channel metrics included ***average mid-channel depth***, ***average mid-channel velocity***, and ***average bank-full width***. All of these flow metrics covered ranges of hydrologic conditions including wetting of habitat at low flows through to potential disturbance of habitat at highest flows.

Chemical analytes in water potentially influenced patterns of accrual by periphyton assemblages mainly through control of nutrient-limited growth. As in previous years, water samples from which analytes were measured were collected at the start and end of the sampling time series at each station (i.e., in 2018 these dates were 4 Oct and 4 Dec). ***Conductivity*** and ***pH*** were measured on each day of water sampling using a WTW model 340i instrument (Weilheim Germany) following calibration in fresh standards and buffers prior to measurement. Water was collected using standard grab methods with the open end of the sampling container oriented upstream of the sampler's hand to avoid sample contamination. Dissolved fractions were filtered in the field immediately after collection. All samples were kept cold and shipped to ALS Environmental in Burnaby, B.C. for analysis of ***ammonium*** (NH_4-N) ***nitrate*** (NO_3-N), ***soluble reactive phosphorus*** (SRP), ***total dissolved phosphorus*** (TDP), ***total phosphorus*** (TP), ***total alkalinity***, ***total dissolved solids (TDS) concentration***, and ***pH*** using procedures in APHA (2006). The lab measurement of pH provided a backup to the field measurement using the WTW. The sum of NH_4-N and NO_3-N was called dissolved inorganic N ***(DIN)*** that is the combination of N species that can be taken up by biota and used in photosynthetic production.

Recruitment of benthic invertebrates to reaches of a river to supply assemblages growing in available habitat can come from drift from upstream and from adult flight and oviposition from other streams or downstream reaches of the same stream. Given that a dam can limit invertebrate drift from upstream (Standford et al. 1996, Marchant and Hehir 2002), two variables were calculated to define recruitment opportunity. One was ***distance from origin***, which was the length of river from source to sampling site. The other was ***distance from the dam***. Methods for these measurements are provided in Table 2.2.

A direct source of recruitment via invertebrate drift to Reach 2 was the Yalakom River. Sampling sites upstream of the confluence of the Lower Bridge River and the Yalakom River would not receive recruitment from the Yalakom and sites in Reach 2, downstream of the confluence would be influenced by recruitment from the Yalakom. A binomial factor was used to identify these differences in statistical analyses to examine links between biological assemblages and habitat attributes, wherein ***categorical on/off coding for potential drift from the Yalakom*** was considered a habitat attribute: 1 was assigned to sites downstream of the confluence and 0 was assigned to sites upstream of the confluence.

Variation in substrata texture and particle size may change habitat spaces between substrata particles and influence availability of food for invertebrates (e.g. periphyton), invertebrate competition, and invertebrate predation (e.g. Mackay 1992, Gore 1982, Tronstad et al. 2007). The use of basket samplers among all trials produced a standard particle size inhabited by benthic invertebrates. This standardization reduced variance associated with particle size. Consequently, no annual quantitative measurement of particle size distribution, by site, was conducted among trials to include as a variable for explaining patterns in benthic invertebrate assemblages.

A potential biological driver variable was relative abundance 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 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. The beginning of the fall sampler incubations overlapped with or immediately followed the timing of pink salmon spawning, which meant there was a pathway for direct effects of salmon-derived nutrients to the invertebrate community during the fall sampling.

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 (Trial 0, 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, November, and December) at each index site to support analysis of spatial and temporal patterns of fish size (as a surrogate for growth information). However, fish sampling in the LBR has generally not been possible during periods when flows are >15 $m^3 \cdot s^{-1}$ due to crew safety concerns, poor catchability, etc., or when water temperatures are <5°C. Further, sample timing has varied across years, there have been changes in emergence timing for fry due to the effects of the flow release, and the high flows in recent years may have a differential effect on survival or displacement out of the study area for juvenile fish according to size. Each of these factors affects the size and number of fish available during a given monthly sample session and are confounding to an understanding of differences or changes in growth. For these reasons, the monthly fish growth data have not been included in any of the analyses to-date.

A recommendation was made in the Year 6 (2017) report to consider eliminating the monthly fish growth sampling and re-allocate that portion of the budget to other activities that would benefit the analyses and results interpretation for this program (e.g., extension of periphyton and benthos sampling to the Yalakom River, continuous turbidity measurement using loggers at various stations in the LBR). This recommendation was accepted by St'at'imc Eco-Resources (SER) and BC Hydro starting with the Year 7 (2018) data collection. As a result, the analysis of fish size among flow treatments included in this report is based solely on fish sampled during the annual stock assessment (see description of this method under "Abundance and Biomass", below). During this task, a sufficient sample size ($n \geq 30$) of each target species and age class has been more consistently acquired for each reach, and the sample timing has been very consistent across years. Forklength (mm) and weight (g) was recorded for each captured fish.

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 habitat surveys that were conducted in reaches 2 and 3 in 1993, and in Reach 4 in 2000 (after initiation of the flow release re-wetted that reach) that inventoried all major meso-habitat types (e.g., runs, riffles, pools).

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 additional sites were added to the upper region of Reach 3 in 1998, bringing the total number of sites for that reach to 20. In 2000, an additional 12 sites were selected in the rewetted Reach 4 by the same procedure that was used for reaches 2 and 3, bringing the total number of sites for all three study reaches to 50. Starting in Trial 2, 1 – 2 sites in Reach 2 were dropped, reducing the number for that reach to 16 or 17 and the total to 48 or 49 since 2012 (Figure 2.1).

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

Table 2.3 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	Transition Year [a]	50	30 Aug – 10 Sep
2001	Trial 1 ($3\ m^3 \cdot s^{-1}$)	50	27 Aug – 10 Sep
2002		50	28 Aug – 5 Sep
2003		50	2 – 11 Sep
2004		50	7 – 15 Sep
2005		50	6 – 16 Sep
2006		50	5 – 14 Sep
2007		50	5 – 19 Sep
2008		50	3 – 18 Sep
2009		49	8 – 24 Sep
2010		50	7 Sep – 19 Oct [b]
2011	Trial 2 ($6\ m^3 \cdot s^{-1}$)	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
2018		49	4 – 15 Sep

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

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

During the Pre-flow period, nets were used to block off the full width of the stream in Reach 3; therefore, the sampled areas included the entire channel. This was not possible in 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 three high flow years to-date (2016 to 2018; see September period on Figure 1.3).

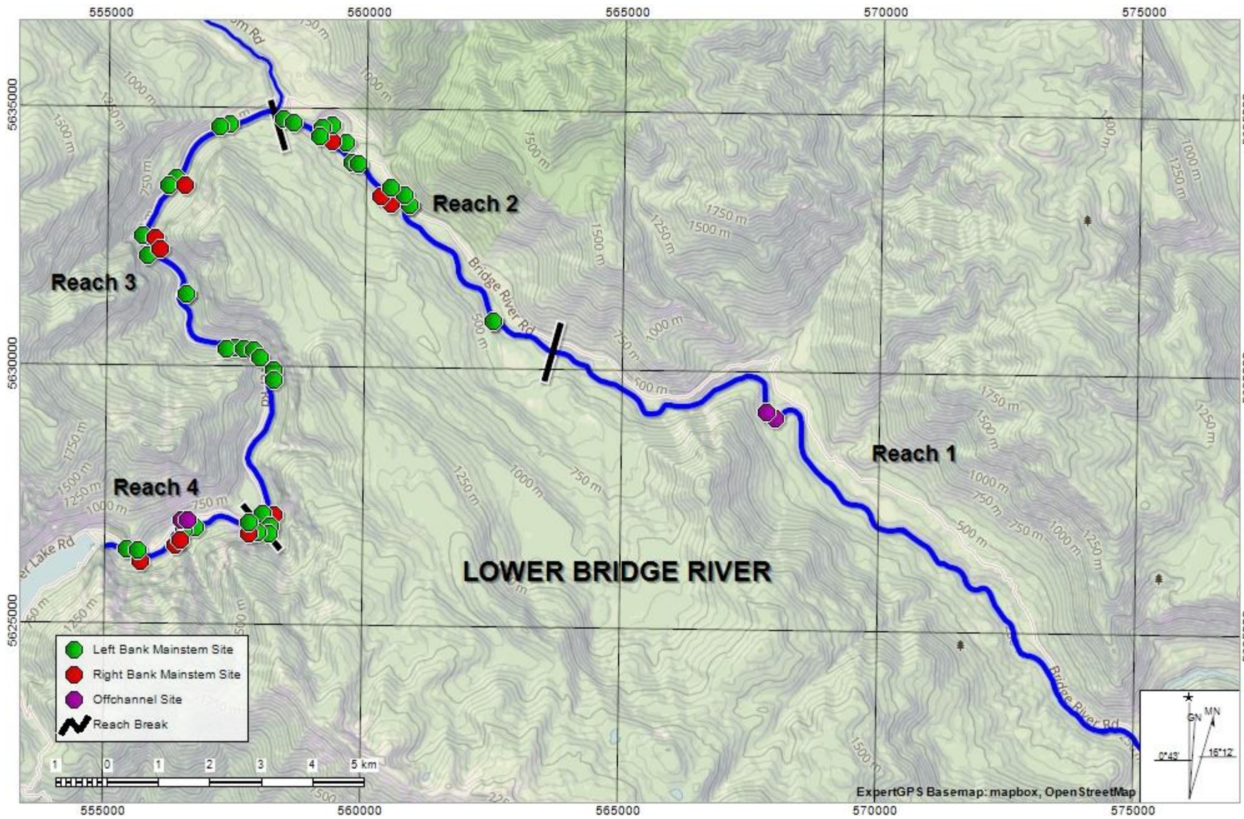


Figure 2.1 Distribution of Juvenile Stock Assessment Sites in the Lower Bridge River study area.

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).

Enhanced Off-channel Sites

Enhanced off-channel sites were sampled for juvenile salmonids for the first time in fall of 2018. A single site in riffle habitat, and a single site in pool habitat was sampled at both the Applesprings enhanced off-channel located in Reach 1, and the enhanced Bluenose off-channel in Reach 4. Fish sampling methods applied at these locations were the same as those used during the mainstem fall standing stock assessment (described above). A multi-pass depletion approach was used to estimate abundance, density, and biomass. Estimates of density and biomass were compared to averages from mainstem sites during 2018 and the average across trials 1 and 2. The total area of each habitat type (riffle and pool) was multiplied by the estimated densities and summed to determine the total abundance for each off-channel. We compared the densities and abundance from these enhanced off-channels, relative to the densities and total abundance in the mainstem, to assess use of these habitats following the high flow period as potential refuge areas, and determine the extent to which they potentially mitigate impacts of high flows. Note: the abundance estimates for the off-channel sites were not included in the total estimates for the mainstem reaches presented in this report in order to maintain consistency with the results and analysis from previous years and flow treatments.

2.1.6. Adult Escapement

Adult spawner count data for the Lower Bridge River (up to 2017) 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 from 2016 to 2018. 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 (streamwalks) conducted under the BRGMON-3 program (2012 to the present) followed methods used in previous years, where two observers walked in a downstream direction on the riverbank and recorded species and location. Viewing conditions, cloud cover, and lateral water visibility were also recorded (Ramos-Espinoza et al. 2018). Starting in 2018, the streamwalks for spawner enumeration were expanded to include spot counts in accessible sections of reaches 1 and 2 (White et al. 2019 in draft). However, since this report only references the spawner estimates up to 2017 (which recruited the juveniles sampled in 2018), the estimates based on 2018 activities have not been included here.

Visual counts occurred weekly for chinook and coho salmon in Reaches 3 and 4. In 2017, surveys started on August 18 for the salmon species, and continued until December 8 when fish activity ceased based on streamwalk and telemetry data. 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, including marked fish, which have been conducted since 2011 (Ramos-Espinoza et al. 2018). However, as noted by the authors, generating accurate and precise AUC estimates from the historic data was hampered by inconsistent sampling methodology and survey area across flow treatments, and a lack of historic observer efficiency data. A key assumption in AUC estimates is that the mean observer efficiency documented by the BRGMON-3 program reflects conditions both before and after the flow release. It is likely that observer efficiency prior to the flow release was higher owing to lower and clearer flows. Thus, escapements prior to the flow release are likely overestimated due to this assumption.

A fish enumeration facility (resistivity counter) was constructed by IFR in October 2013 near the downstream end of Reach 3 to obtain more precise escapement estimates for coho, chinook and steelhead above the Yalakom confluence going forward. Based on results in other systems, resistivity counters can provide accurate estimates (with confidence limits +/- 10% of true abundance). In future, these counter-based estimates can be compared to the estimates based on visual methods as a means of calibrating the historic estimates (though such a comparison would only apply to post-flow release counting conditions and would not address the bias described in the preceding paragraph). However, at the time of this report, only a few years of data from the resistivity counter were available (Ramos-Espinoza et al. 2018).

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 (Ramos-Espinoza et al. 2018).

2.2. Modified Operations (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 additional impacts as well. To address some of the identified effects in the immediate-term, additional high flow monitoring was incorporated to supplement the BRGMON-1 program. This work was implemented under three new high flow monitoring programs for modified operations from Terzaghi Dam: 1) High flow monitoring of water quality parameters, kokanee entrainment, bank erosion and deposition, and potential fish stranding locations; 2) Surveys to document juvenile salmonid habitat use under high flows; and 3) High flow ramp down monitoring and stranding risk assessment. The following information was largely summarized from the LBR High Flow Monitoring Field Report produced by Coldstream Ecology, Ltd. (O'Farrell and McHugh 2018).

2.2.1. High Flow Monitoring

Water Quality

The high flow period in 2018, defined as the period when flows exceeded the Trial 2 peak (15 $m^3 \cdot s^{-1}$), was between 10 May and 1 August 2018 (83 days). Water quality monitoring in 2018 was conducted on 20 dates between 6 May and 26 July 2018. Monitoring on dates prior to the start of high flows was to document baseline conditions. As in 2016 and 2017, high flow monitoring included spot measurement of air temperatures, water temperatures, total dissolved gases (TDG), 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 (river km 40.5), Russell Springs (Rkm 30.4), and just upstream of the Yalakom River confluence (Rkm 25.0).

Spot measurements were taken 2x per week from 6 May to 9 July, which spanned the ascending limb and peak of the high flow hydrograph. From 10 to 26 July, across the descending limb, measurements were taken 1x per week. An additional pre-season baseline datapoint for turbidity was measured on 1 March 2018 from all three sampling locations, and turbidity measurements in the Yalakom River were added to the weekly monitoring on 29 May 2018 for additional reference data. Conductivity and alkalinity measurements were added on 13 July 2018 at the request of Rich McCleary, Ministry of Forests, Lands, Natural Resource Operations and Rural Development (FLNRORD).

Spot water temperature measurements were taken at >60 cm depth in the water column, and TDG and turbidity monitoring were conducted according to BC Hydro protocols, as specified in the 2018 Lower Bridge River High Flow Monitoring Scope of Services. To facilitate comparability of results, the same time of day was targeted on each sampling day for measurements. A Point Four Tracker portable total gas pressure meter was used to measure TDG and water temperature. Barometric pressure was measured on each sampling day for calibrating the instrument. Turbidity samples were collected by plunging a clean sample bottle (that had been pre-rinsed 3x with river water) below the surface until full. Turbidity (as Nephelometric Turbidity Units, or NTUs) of each sample was measured by a LaMotte 2020we portable turbidity meter.

Kokanee Entrainment

To assess the incidence of kokanee entrainment from Carpenter Reservoir into the Lower Bridge River channel during the period of high flows (>15 $m^3 \cdot s^{-1}$), visual streamwalks were conducted to observe and enumerate kokanee (live and mortalities). The surveys were conducted by two technicians from Coldstream Ecology Ltd. and Xwísten on the same 20 dates as the water quality measurements. 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.

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

The data from these surveys were intended to: a) document whether kokanee entrainment occurred during the 2018 high flow event; b) establish an index of the number observed on the survey dates; and, c) record some observations about the fish that were found. Based on these data, it is not possible to estimate total numbers of entrained fish (since observer efficiency was not assessed—which would require a much larger and more complex monitoring approach), determine the proportion of entrained fish that were live or mortalities, or determine the specific cause of the observed mortalities.

Bank Erosion and Sediment Recruitment

Crews also conducted surveys to identify and assess bank erosion and sediment recruitment sites associated with the high flows. A set of fourteen bank erosion and deposition sites in reaches 2, 3 and 4 that had been identified by surveys conducted in 2017 by Kerr Wood Leidal Associates Ltd. (KWL; Ellis et al. 2018) were re-visited for repeat observations. In addition to the pre-selected sites, surveys also included documentation of any new erosion or deposition areas that were observed. However, it was noted in Coldstream's summary field report that the "volume of deposition at each site could not be estimated and monitored due to high flows" (O'Farrell and McHugh 2018). The study reaches were surveyed weekly from 6 May to 26 July 2018, by vehicle and shore-based streamwalks.

For each identified erosion area, recorded parameters included: site name or location description, GPS coordinate, river bank, reach, dimensions (approx. length and width in meters; area in m^2), flow release discharge ($m^3 \cdot s^{-1}$) when initially observed, substrate composition (% by size class), and material origin (road, natural). All sites, with the exception of the Fraser Lake rip rap site (approx. Rkm 33.2) were assessed using binoculars at distances of 30 m to 300 m. Photographs were taken at each location for reference (only a few are included in this report; the remainder can be provided upon request).

Fish Stranding Site Reconnaissance

Field reconnaissance during the high flow (>15 $m^3 \cdot s^{-1}$) period identified potential new fish stranding sites in reaches 1, 2, 3, and 4 and qualitatively assessed known strand areas for risk at the range of 2018 high flows. The reconnaissance was useful for proactively guiding fish salvage efforts during the high flow rampdown events. Surveys were conducted between 11 May and 26 July 2018, and the survey frequency was dictated by the extent of flow change and habitat area flooded or dewatered since the last observation. 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.

For each identified site with fish stranding potential, crews assigned a low, medium or high stranding risk to each site, given the available habitat information. Stranding risk was assigned 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.

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

2.2.2. Juvenile Salmonid Habitat Availability and Displacement

Starting in 2018, fish sampling was undertaken during the high flow period in the LBR mainstem to document potential refuge habitats along the river margins, assess the relative use of these areas by juvenile salmonids, and potentially contribute to an understanding of fish displacement within the study area. This component was a pilot study in 2018 with the goal of informing future study designs. By repeat-sampling the same set of sites across a range of high flows, the intent was to assess changes in use of those sites by juvenile salmonids across the ascending and descending limbs of the 2018 hydrograph, measure depths and velocities to characterize those habitats, and document the sample-able area of the sites at each flow.

Surveys were conducted on 18 dates between 9 May and 30 July 2018, at 7 different flow release discharges. The first survey targeted peak flows for the Trial 2 hydrograph (i.e., 15 $m^3 \cdot s^{-1}$), which were considered 'baseline' data for comparison with the high flow results. However, only two sites were sampled for this survey (Bluenose Outflow and Applesprings Outflow). The other surveys were conducted at three discharges on the ascending limb of the high flow hydrograph (28, 70 and 100 $m^3 \cdot s^{-1}$), and three on the descending limb (82, 67 and 27 $m^3 \cdot s^{-1}$).

Site locations were pre-selected by BC Hydro staff based on a modelling exercise (using BC Hydro's Telemac2D model) that predicted potential rearing habitat areas according to Habitat Suitability Index criteria across a range of high flows. Ten sites were distributed within reaches 4 and 3: Nine of these were mainstem sites (5 considered high quality habitat and 4 considered low quality), and one was in the off-channel habitat at Bluenose (Rkm 39.2 – considered high quality) (Table 2.4). In addition to these ten sites, three sites were established in Reach 1 within the Applesprings off-channel habitat (high quality), and two in the LBR mainstem just upstream of the Applesprings habitat and at the Applesprings outflow channel (also considered high quality).

Table 2.4 Summary of sites sampled for the juvenile salmonid habitat use surveys at high flows.

Reach	Site	Approx. Rkm	Habitat Quality	Description
4	Eagle Lake	39.5	High Quality	Mainstem
			Low Quality	Mainstem
	Bluenose Offchannel	39.2	High Quality	Offchannel
	Bluenose Outflow		High Quality	Mainstem
3	AE Rkm 34.0	34.0	High Quality	Mainstem
			Low Quality	Mainstem
	Near Russel (U/S)	30.4	High Quality	Mainstem
	Near Russel (D/S)		Low Quality	Mainstem
	Boundary	29.0	High Quality	Mainstem
			Low Quality	Mainstem
1	LBR Mainstem at Applesprings	11.3	High Quality	Mainstem
	Applesprings Upper	11.2	High Quality	Offchannel
	Applesprings Middle		High Quality	Offchannel
	Applesprings Lower		High Quality	Offchannel
	Applesprings Outflow		High Quality	Mainstem

Selected sites were sampled in an upstream direction along one bank (for mainstem sites) or bank-to-bank (for offchannel sites) by open-site electrofishing. As such, catch results for these surveys represent a minimum estimate at the time of each survey. Juvenile (Age-0+ and Age-1) coho, chinook, and *O. mykiss* were the target species and age classes. Site length or area varied depending on the amount of habitat available at the selected sites under the flow conditions at the time of each survey.

All fish collected during sampling were identified to species and age class (estimated), measured for length and weight, and a sub-set were photographed. Electrofishing effort (seconds) and the number of crew members carrying out sampling was recorded. Following fish processing, flow measurement transects were completed for the sampled extent of each site. These data were intended for calibrating the predicted depths and velocities according to the hydrodynamic HSI model of the site (to be carried out in a separate scope of work). The locations of the upper and lower extent of each site were recorded with a GPS, the length of shoreline, area sampled, and the general characteristics of the site (habitat type, dominant/sub-dominant substrate, water visibility) were also recorded. Representative photographs of each site were taken, as well as general information (time of sampling, conditions, discharge level, etc.).

2.2.3. High Flow Ramp Down Monitoring and Stranding Risk Assessment

Flow ramping and fish salvage data were collected as part of High Flow Ramp Down Monitoring at LBR discharges >15 $m^3 \cdot s^{-1}$, and across the range of WUP Trial 2 flows (15 to 1.5 $m^3 \cdot s^{-1}$). The methods described in this section come from 2018 weekly monitoring reports and documentation provided by Coldstream Ecology Ltd. (O'Farrell and McHugh 2018).

2018 discharge data for Terzaghi Dam and river stage data for Rkm 36.8 (~4 km downstream from the dam; a.k.a. the compliance location) were provided by BC Hydro Power Records. The data were available as hourly values.

On each ramping date before any flow changes were initiated, field reconnaissance of the survey area was completed at an overview level to identify and rank specific locations with potential fish stranding risk, 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.

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

As per the BC Hydro Bridge-Seton Fish Stranding Protocol, fish salvage types were defined as follows:

Incidental: fish in habitats that are not yet isolated, and still have the opportunity to move to deeper areas on their own (applied when water temperatures were too warm);
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 2018 were compared and combined with the 2016 and 2017 data, the only other years where high flow ramp downs have occurred since the start of the continuous flow release, 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 $m^3 \cdot s^{-1}$ (i.e., WUP Trial 2 flows). 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.

2.3. Data Analysis

2.3.1. Benthic Communities

Layout

The management question associated with benthic communities (Question 2, Section **Error! Reference source not found.**.1) states, "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?" The benthic assemblages provide an indication of overall biological condition of the river and the invertebrates show food available for fish. Given that rates of production among primary producers (benthic algae) and secondary producers (benthic invertebrates) have never been measured or described in Terms of Reference documents for BRGMON-1, the term "production" in the management question is considered something that is an indicator of production. True production requires a rate term that is absent in all data.

The assigned measurement in all river benthic monitoring for BRGMON-1 is abundance or biomass of cells or animals per unit area by taxon. For periphyton, the biomass metric was the amount of chlorophyll-a per unit area, where chlorophyll-a is a plant pigment that is a measure of biomass of photosynthetic algae. This measure is preferred over something like dry weight that can include non-biological material in the stream substratum or organic content that can include non-photosynthetic biomass (e.g. bacteria, fungi, detritus of terrestrial origin). Periphyton assemblages were defined by cell density (number of cells per unit area) measured for each taxon. Benthic invertebrates were quantified as counts of animals in growth stages that were actually growing in the water (mainly larval forms of aquatic insects). Individuals from terrestrial habitats or adults of aquatic insects were not included in the animal counts. All invertebrate data were expressed as number of individuals per basket sampler or per unit area where the planar dimension of the basket lying on the stream substratum was the area of sample. Biomass of the benthic invertebrates was not measured.

We expected characteristics of place in the river to modify the patterns of biological assemblages as well as variation in flow associated with Trial because of differences in habitat and different recruitment of biological assemblages between locations. Some of those characteristics are listed in Table 2.2. Place was defined as Reach of river. We were not particularly interested in specific locations defined by sampling site because site is not expected to be used for making management decisions. We were interested in the different reaches because there are clear physical separations between them, and they are commonly referred to in discussions about flow decisions. Reach 4 was dewatered during Trial 0 and is closest to the dam, making it unique. Reach 3 has always been wetted but does not receive benthos recruitment from unimpounded headwaters as does Reach 2. These distinctions made Reach an important factor.

In this layout, the mean values of measurements (e.g. invertebrate counts by taxon) collected from individual samplers within sites and Reaches within a year were calculated for space (Reach) x time (Trial) comparisons. Those multiple samples within a Reach added precision to the observations within a year but were not "replicates" for statistical inference. They were pseudoreplicates with respect to testing a flow effect, which is invalid (Hurlbert 1984, Stewart-Oaten et al. 1986, Underwood 1994). The group of samples within a year was the experimental unit because any one year could receive a different treatment (i.e. a flow treatment). A place in the river was not the experimental unit because that place cannot receive a flow treatment that is different from that applied at another place. To be statistically correct, the flow trials would be randomized among years, but randomization was not practically possible. It was also not particularly important because the differences in flows among trials were large. From a water management point of view, if differences among benthic assemblages associated with Trial did not show up with the large differences in flow release between trials, Trial would not be considered important in modifying benthic assemblages.

The absence of a control reach that could be sampled like the Lower Bridge River but not receive the flow Trials prevented a test of flow effect on the benthic assemblages because temporal change not related to flow could not be measured. We were able to compare the invertebrate assemblages over space (place on the Lower Bridge River) and time (years blocked by Trial) but this was not a test of a "treatment effect" (i.e. flow) on those assemblages. It was only a test of a place effect defined by Reach and a time effect defined by Trial with interactions of place (Reach) and time (Trial).

The calculation of mean values of a given endpoint (i.e. invertebrate counts by taxon) for every combination of Reach and Trial among years resulted in 57 experimental units for analysis of Reach and Trial effects (Table 2.5). A total of 406 individual samples contributed to this layout of experimental units including the loss of 5 samples during field activities (1.2% loss rate).

Table 2.5 Layout of factors, experimental unit, and number of samples for the benthic invertebrate analysis.

<table>
<tr><th>Description of factors, experimental unit, and samples</th><th>Trial factor</th><th>Reach factor</th><th>Years (experimental unit replicated)</th><th>Mean of these samples contributed to each yearly observation</th><th>Number of samples (number of years x sites x samples)</th></tr>
<tr><td rowspan="12">Factor levels and number of observations</td><td rowspan="3">Trial 0</td><td>Reach 4</td><td>Dry substrata – no samples</td><td>Dry substrata – no samples</td><td>0</td></tr>
<tr><td>Reach 3</td><td>3</td><td>4 sites x 3 samples = 12 (less one missing)</td><td>35</td></tr>
<tr><td>Reach 2</td><td>3</td><td>2 sites x 3 samples = 6</td><td>18</td></tr>
<tr><td rowspan="3">Trial 1</td><td>Reach 4</td><td>9</td><td>1 site x 3 samples = 3 (less one missing)</td><td>26</td></tr>
<tr><td>Reach 3</td><td>9</td><td>4 sites x 3 samples = 12 (less one missing)</td><td>107</td></tr>
<tr><td>Reach 2</td><td>9</td><td>2 sites x 3 samples = 6</td><td>54</td></tr>
<tr><td rowspan="3">Trial 2</td><td>Reach 4</td><td>5</td><td>1 site x 3 samples = 3</td><td>15</td></tr>
<tr><td>Reach 3</td><td>5</td><td>4 sites x 3 samples = 12 (less 2 missing)</td><td>58</td></tr>
<tr><td>Reach 2</td><td>5</td><td>2 sites x 3 samples = 6</td><td>30</td></tr>
<tr><td rowspan="3">Trial 3</td><td>Reach 4</td><td>3</td><td>1 site x 3 samples = 3</td><td>9</td></tr>
<tr><td>Reach 3</td><td>3</td><td>4 sites x 3 samples = 12</td><td>36</td></tr>
<tr><td>Reach 2</td><td>3</td><td>2 sites x 3 samples = 6</td><td>18</td></tr>
<tr><td colspan="3">Total Count</td><td>57</td><td></td><td>406</td></tr>
</table>

In this analysis of benthic communities using data from 1996 through 2018, seasons of measurement (spring, summer, fall) that were shown in the 2017 analysis are limited to the fall period. This selection is appropriate because benthos was only sampled in the fall in 2018, consistent with all of the previous Trial 3 sampling episodes. Data from spring and summer that was only collected during Trial 0 and 1 was presented in last year's report (Sneep et al. 2018). It has not changed and there is no need to repeat it here.

Periphyton

The periphyton data analysis was descriptive to show the community supporting invertebrates that graze on algae and collect drifting algal particles. Stacked bar graphs showed cell counts of main algal classes among Trials in each of the three reaches. Common species within the prevalent classes were identified. Mean PB ± SD was calculated to show change in this index of algal production by Trial and place in the river defined by sampling station. This PB data was used as a habitat attribute in analyses linking benthic invertebrate assemblages to habitat conditions, as explained in methods for benthic invertebrate analysis (next section).

Benthic Invertebrates

The first analysis of benthic invertebrates was to produce stacked bar graphs showing density by invertebrate class in all trials and reaches. This overall view of the assemblages provided a high level perspective of changes over space and time. Invertebrate densities measured in the Yalakom River were included for comparison although they were not used in any Trial x Reach statistical tests because the data were only from one year.

The first analysis of benthic invertebrates was a 2-way crossed PERMANOVA (Anderson et al. 2008, Oksanen et al. 2017) used to examine Trial and Reach effects on the assemblages of invertebrate families. The significant probability level was set at p=0.05. Family level counts were selected because not all taxa were identified to lowest levels and because family level data are equally sensitive to lower level identifications for distinguishing environmental change (Bowman and Bailey 1998, Arscott et al. 2006). The PERMANOVA is the multivariate analogue of a univariate (e.g. count of a single taxon) 2-way analysis of variance (ANOVA). Our data was multivariate because an invertebrate family was a variable and there were many of them (57 families at last count). The two factors were Trial with 4 levels (Trials 0, 1, 2, 3) and Reach with 3 levels (Reaches 4, 3, 2). The experimental unit was a year of samples from sites coded by Reach and Trial as shown in Table 2.5. Years were not pseudo-replicates because we were not inferring a treatment (i.e. flow) effect from the tests of significance in PERMANOVA. The error term applied only to variance in assemblages by Trial and Reach and interactions therein.

Sampling between years was sufficiently well separated in time such that assemblages sampled at a given station in say Year 2 were not influenced by assemblages at that same station in Year 1. Most of the invertebrates have life cycles much shorter than a year and they drift downstream as part of habitation in a stream. These behaviours mean that an assemblage in one year was not affected by an assemblage in a previous year. This independence satisfied requirements for independent experimental units for analysis of Trial and Reach effects in PERMANOVA or any kind of multifactorial design (Hurlbert 1984).

To determine which families contributed most to dissimilarities among invertebrate assemblages among Trials and Reaches, a multivariate similarity percentages (SIMPER) procedure with Bray-Curtis dissimilarities was used (Clarke and Gorley 2015). This procedure calculates the overall contribution of each family to the Bray-Curtis dissimilarity. Families cumulatively contributing 70% of the dissimilarities between Trials and Reaches were considered indicators of effects of Trial and Reach. Line graphs were used to show change in these indicator taxa as well as total invertebrate abundance and a diversity metric called family (count of families in a sample) among combinations of Trial and Reach.

A significant interaction of Trial and Reach in the PERMANOVA ($p<0.05$) showed that the invertebrate assemblages changed differently among reaches in the different trials. It would mean that the individual factors could not be examined for an independent statistical effect because each one was confounded by the other. In such a case, the change in abundance of indicator taxa was plotted by Trial and Reach to show the interaction. For clarity on the figure, error bars were not applied because the interaction was already shown to be statistically significant. If the interaction was not significant, the individual factors (Trial and Reach) were examined as follows. A significant Trial effect meant that some aspect of habitat (variables including flow listed in Table 2.2) was driving the time course change. If Trial was not significant, it meant that changing habitat conditions over time including the change in flows associated with Trial were not affecting the invertebrate assemblages. Similarly, a significant Reach effect would show that the invertebrate assemblages changed between the three different reaches, independent of effects of Trial. If Reach was not significant, it meant that changing habitat conditions over space did not affect the invertebrate assemblages.

We used redundancy analysis (RDA) to determine the response of benthic invertebrates to changing values of habitat attributes of possible importance (Table 2.2) using the complete history of data from the fall sampling periods beginning in 1996. The RDA showed what habitat conditions contributed to variation in benthic invertebrate assemblages over space and time. Results supported the PERMANOVA in explaining what habitat conditions contributed to any significant Trial and Reach effects and their interactions on the invertebrate assemblages. Remember the PERMANOVA cannot show effect of flow; it is only a test of space and time effects. The RDA showed the relative importance of the habitat attributes in modifying assemblage patterns shown in the PERMANOVA. The same 57 experimental units used in the PERMANOVA (Table 2.5) were used in the RDA.

Redundancy analysis is analogous to multiple linear regression followed by principal component analysis (PCA) (Legendre and Legendre, 2012a). RDA estimates the amount of variation in a standardized total family abundance data matrix **Y** that is explained by the standardized matrix of ecological variables, **X**. **Y** was laid out as a fourth root transformed matrix of invertebrate family counts as recommended by Anderson et al. (2008) and **X** was a matrix of habitat variables values. The X matrix data was standardized to correct for different units of measurement among variables. **X** and **Y** had identical experimental units (named as samples in a year; **Error! Reference source not found.**).

The habitat variables were those that may directly affect assemblages of benthic invertebrates and were not correlated with each other. Without using statistics, we shortened the starting list of variables in Table 2.2 to one with fewer variables that were not ecologically correlated. Variables were excluded if: a) they were redundant with other variables; b) they did not vary among stations or over time; or, c) they would not affect assemblages in the samplers. That process resulted in a list of 13 variables shown in Table 2.6. Plots of each pair of these variables followed by calculation of a correlation matrix was used to test for co-linearity of these variables. Correlations greater than 60% were considered too high for the variables to be independent. When these high correlations occurred, the variable having the most ecologically meaningful link to the invertebrate assemblages was selected for the RDA and the other variable was deleted from analysis.

Table 2.6 List of habitat variables included in the benthic invertebrate RDA.

Variable name	Rationale for including in RDA
Peak Biomass (PB)	***Included*** as a measure of autotrophic food production for benthic invertebrates
Mean water temperature	***Included*** as a measure of physiological limit to invertebrate growth
Incubation flow	***Included*** as a measure of flow to which the samplers were exposed
Disturbance flow	***Included*** because it describes the flow associated with the flow trial preceding benthos sampling
Peak disturbance flow	***Included*** as an indicator of maximum energy affecting the Lower Bridge River in a given Trial
Mean depth at sampler	***Included*** because it is a specific water depth at the sampler that may be independent of flow because it is sensitive to where the sampler is placed
Mean velocity at sampler	***Included*** because it is a specific water velocity at the sampler that may be independent of flow because it is sensitive to where the sampler is placed
Mean DIN concentration	***Included*** because N is a nutrient that may limit periphyton growth that supplies food for invertebrates. Values may be independent of PB if PB had large variance masking its importance as an indicator of food supply. In such a case DIN may be that indicator. Potential correlation of DIN with PB required testing before it could be used in RDA modeling.

Variable name	Rationale for including in RDA
Mean TDP concentration	***Included*** because TDP is a form of soluble phosphorus that may limit periphyton growth that supplies food for invertebrates. Values may be independent of PB if PB had large variance masking its importance as an indicator of food supply. In such a case TDP may be that indicator. Potential correlation of TDP with PB required testing before it could be used in RDA modeling.
Distance from origin	***Included*** as a surrogate measure of potential recruitment of benthos from upstream
Distance from dam	***Included*** as a surrogate measure of potential recruitment of benthos from upstream
Yalakom influence	***Included*** as a surrogate measure of potential recruitment of benthos from upstream to Reach 2 and not to Reaches 3 and 4
Pink run in on or off year	***Included*** as a measure of added nutrient supply for autotrophic production that affects food for benthic invertebrates and a supply of particulate organic matter for feeding by some groups of aquatic insects. This effect would be greater in an on pink year than in an off pink year.

The RDA was run using the dbRDA algorithm in Primer v7 (Anderson et al. 2008). Output first showed how much variance in the biological assemblages was explained by each included habitat variable (Table 2.6) in the absence of any other variable. This step was largely exploratory to examine dominant and less dominant habitat attributes driving assemblage patterns. Very weak explanatory variables were eliminated at this stage. We next used the "Best" procedure in dbRDA to report a series of models beginning with the best 1 variable model, best 2 variable model, etc. up to a model including all variables. Selection of what was a "Best" model was based on review of adjusted R^2 (proportion of explained variation in the model adjusted for the number of parameters in the model) and AIC (Akaike 1973). The selected model had highest R^2 and lowest AIC with fewest habitat variables. Significance was tested by permutation 999 times (Clarke and Gorley 2015).

A constrained distance-based ordination, called a dbRDA figure, was drawn of the predictor variables in Primer v7 to visualize the selected model (Anderson et al. 2008). The ordination was constrained to several axes (although only two typically explain most of the variance), each showing variation fitted to the model and total variation. This ordination provided another way to interpret the adequacy of the model. A model was considered adequate if the fitted variation (explained by the model) along any one axis was a large proportion of total variation (explained and unexplained by the model) along that same axis. If fitted variation only explained a small amount of total variation, the model was considered weak. This outcome would show that much variance was unexplained by the model. Given that flow is part of the suite of explanatory variables, this outcome would show weak effect of flow on the temporal and spatial patterns of invertebrate assemblages.

The dbRDA figure showed the habitat variables represented by vectors or arrows; longer arrows showed more correlation with the redundancy axes and therefore were more related to the variation in the biological assemblage data matrix (Ter Braak 1987; Legendre and Legendre, 2012b). Symbols were used to identify the relationship between the habitat variables (the arrows) and the family assemblages (sites shown by the symbols) (Ter Braak 1987; Legendre & Legendre 2012a).

2.3.2. Juvenile Fish Production: Size

As conducted in the analyses for the Year 6 (2017) report, we evaluated effects of flow on juvenile salmonid growth based on weight samples taken during the annual fall stock assessment. The only update to the juvenile fish size analysis presented here was the inclusion of the 2018 data as an additional high flow year. The results as presented for previous flow treatments did not change.

Using weight as a surrogate for growth assumes that the interval between emergence date and sampling date are relatively consistent among years, or at least among flow treatments. There was some variation in sampling dates for stock assessment among years, particularly between the first three years of the Pre-flow period (early to mid October from 1996 to 1998) and the subsequent flow treatments (late August to late September from 1999 to 2018; see Table 2.3, above). Generally, the variation within the flow trial years was low. Owing to changes in water temperatures due to differences in flow treatments, emergence timing was likely different, especially for chinook where water temperature differences over the incubation period between the pre-treatment and later flow treatments have been large (Section 3.1.1). Thus, using weight data to make inferences about growth is problematic, especially for chinook. Nevertheless, we computed average weight for each reach and flow treatment, and for the 2016-2018 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.

2.3.3. Juvenile Fish Production: Abundance & Biomass

The abundance and biomass of juvenile salmon in each reach was estimated with a hierarchical Bayesian model (HBM) described in Bradford et al. (2011) and Appendix 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 (Trial 3).

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 2.7.

Table 2.7 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 $m^3 \cdot s^{-1}$	2016-2018	2017-2018

Note that data from 2000 was not used in the average for the Pre-flow or Trial 1 treatments because the change in flow occurred midway through the growing season and it is unclear how juvenile fish (both fry and parr) would have been affected in that year. There was no need to skip a year during the transition from the Trial 1 to Trial 2 treatments because flow changes occurred at the start of the growing season and prior to the emergence of mykiss fry in that year (2011). Despite a higher peak flow in 2015 (i.e., 20 $m^3 \cdot s^{-1}$ instead of 15 $m^3 \cdot s^{-1}$) owing to particular conditions and reservoir management decisions in that year, 2015 was included in the Trial 2 treatment because the yearly average (i.e., 6.6 $m^3 \cdot s^{-1}$) was still very close to the average for other years in this treatment (i.e., 5.3 to 6.1 $m^3 \cdot s^{-1}$). Age-0+ abundance in 2016, 2017 and 2018 were used to compute the average abundance and biomass for the High flow (Trial 3) 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.

2.3.4. Stock-Recruitment Analysis

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

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

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 (Ramos-Espinoza et al. 2018). Escapement estimates for these species represent abundance in reaches 3 and 4 only as this is where the longest time series of stream walks have been conducted. Counts were expanded to estimates of the number present based on estimates of observer efficiency, which were determined from mark-resight data. A normal distribution was fitted to the expanded count data from each year, and the total escapement was determined by dividing the area under the normal curve by the survey life. The escapement estimates for each calendar year were plotted against fry abundance the following calendar year (e.g., chinook spawning in September of 2017 produced fry that were sampled in the fall of 2018). We then fit the following Beverton-Holt model to these data,

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

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

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

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

3. Results

3.1. Core Monitoring Components

3.1.1. Discharge, Water Temperatures and Salmon Incubation

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), including: flow and water temperature. The high flow releases since 2016 have resulted in greatly increased discharges in late spring and early 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, 2017 and 2018 (i.e., 97, 127 and 102 $m^3 \cdot s^{-1}$, respectively) were 6.5-, 8.5- and 6.8-fold higher than typical Trial 2 peak flows (i.e., 15 $m^3 \cdot s^{-1}$), and mean annual flows were 3.6-, 3.1- and 3.0-fold higher than the Trial 2 average (i.e., 6 $m^3 \cdot s^{-1}$), respectively. To-date, the mean flow for Trial 3 years combined (2016-2018) has been 19.5 $m^3 \cdot s^{-1}$.

Due to minimal tributary and groundwater inflows in reaches 4 and 3 relative to the magnitude of the release, site-specific discharge estimates were very similar across those reaches (Site A – 39.9 km to Site E – 26.4 km), differing by a maximum of ~4 $m^3 \cdot s^{-1}$ (or 12%) across that distance in 2018 (Figure 3.1). Due to the contribution of the Yalakom River, site-specific discharge at locations in Reach 2 were up to 35 $m^3 \cdot s^{-1}$ (or 55%) greater than 2018 release flows.

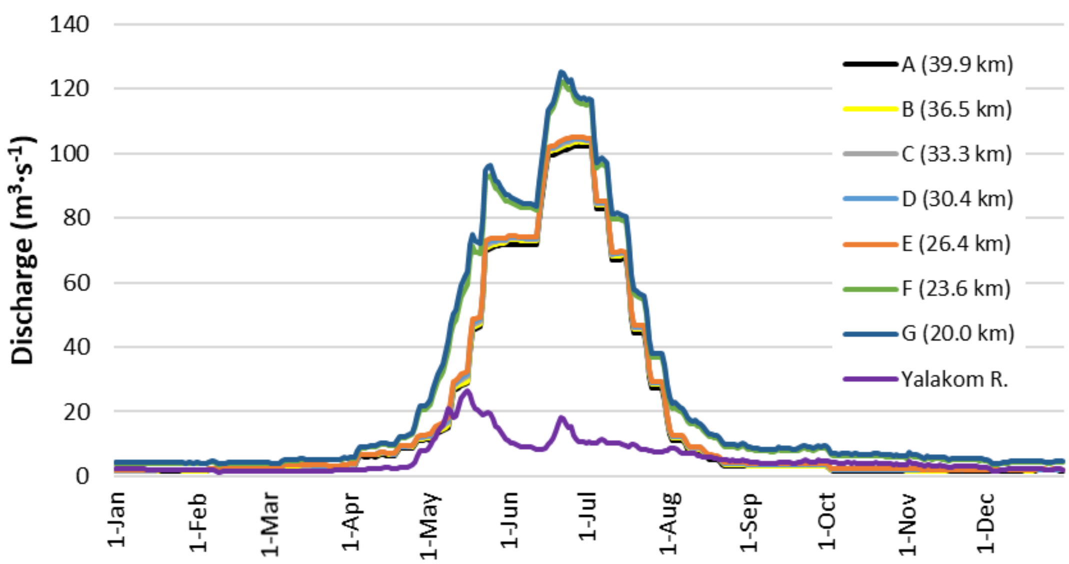


Figure 3.1 Site-specific discharge estimates (shown as mean daily values) in the Lower Bridge River during 2018. Site A is in Reach 4, sites B to E are in Reach 3, and sites F and G are in Reach 2. 2018 Yalakom River discharge is also shown.

The Yalakom hydrograph in 2018 had a primary peak in mid-May, followed by a smaller secondary peak at the end of the third week of June. The ascending limb of the flow release hydrograph (i.e., from 10 May to 15 June 2018) overlapped the timing of the primary Yalakom peak, whereas peak release flows from Terzaghi Dam (i.e., from 16 June to 3 July 2018) coincided more closely with the secondary peak on the Yalakom. The high release flows were ramped back down to Trial 2 peak levels between 4 July and 1 August 2018, and then the releases followed the typical Trial 2 hydrograph shape from August to the end of December.

Relative to the Pre-flow period (Trial 0), dam releases have caused water temperatures to be cooler in the early spring period (Mar-Apr), and warmer throughout the fall (Figure 3.2). These effects were most evident in reaches 4 and 3, with a gradient of effect among stations associated with proximity to the dam. In addition to continuation of these effects, Trial 3 flows from 2016 to 2018 were also characterized by warmer temperatures in January and February, and during the period of the year when the high flows were delivered, typically from May to July. The cause of the warmer water temperatures in January and February are uncertain because the effect was apparent in the Yalakom River (though to a more limited extent than the mainstem LBR; Appendix B), suggesting a potential system-wide ambient temperature effect on tributary inflow temperatures. However, mean monthly air temperatures in nearby Lillooet, BC were the lowest during this period in Trial 3 (Table 3.1; Data provided by Environment Canada).

Table 3.1 Mean monthly air temperatures for Lillooet, BC summarized by LBR flow trial (data provided by Environment Canada).

Flow Trial	Mean Monthly Air Temperatures (± SD)												Trial Average
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	
0	-1.8 (±3.0)	3.4 (±0.2)	5.8 (±1.1)	10.4 (±0.6)	15.4 (±3.0)	19.2 (±1.7)	22.4 (±3.2)	23.0 (±1.1)	17.9 (±1.6)	9.5 (±0.5)	4.6 (±0.5)	0.7 (±1.4)	10.9 (±0.8)
1	-1.6 (±2.0)	1.3 (±1.3)	5.3 (±2.2)	10.3 (±1.3)	15.2 (±1.2)	19.4 (±1.5)	23.4 (±1.2)	22.0 (±0.9)	16.9 (±1.2)	9.7 (±0.9)	2.8 (±2.9)	-2.3 (±2.9)	10.2 (±0.4)
2	-1.1 (±1.5)	1.3 (±3.4)	5.8 (±1.9)	10.0 (±1.0)	15.6 (±1.7)	19.5 (±1.9)	23.5 (±1.9)	22.9 (±0.6)	17.6 (±1.5)	10.5 (±1.3)	2.1 (±0.8)	-0.4 (±0.6)	10.6 (±0.7)
3	-2.2 (±1.6)	0.6 (±2.9)	6.3 (±1.6)	11.3 (±2.4)	17.3 (±1.6)	19.9 (±0.7)	23.2 (±1.4)	22.7 (±1.3)	16.2 (±1.9)	8.9 (±0.7)	4.8 (±2.0)	-2.4 (±3.1)	10.6 (±0.5)

The warmer spring/summer (May-Jul) temperatures in Trial 3 were also evident in all three study reaches, and extended beyond the high flow period in reaches 4 and 3. However, given that water temperatures in the unregulated Yalakom River during this period in Trial 3 were well within the range of temperatures observed for that river during the other flow trials (refer to figures in Appendix B), the differences observed in the mainstem LBR in spring and summer are not likely to be solely due to ambient temperature differences among trials. Rather, the warmer release temperatures in summer may be caused by an effect of the higher conveyance of water through Terzaghi Dam on draw from the various thermal layers in Carpenter Reservoir. The CE-QUAL temperature model developed for Carpenter Reservoir under BRGMON-10 could be consulted to confirm if this is the case. However, the analyses required to determine the cause and mechanism of the increased spring/summer temperatures were beyond the scope of this report.

Water temperatures during the early part of the salmon incubation period in fall (i.e., Sep to Dec) were elevated in Trials 1, 2 and 3 (relative to Trial 0) by up to 4°C at the top of Reach 3. Differences among the three flow treatments during that seasonal period were small, as were differences among years within trials (though release temperatures were slightly higher in 2016 from 1 November to 9 December of that year – refer to Reach 4 figure in Appendix B). Release flows among the three flow trials have been very similar across the fall period (3.0 $m^3 \cdot s^{-1}$ in Sep; and between 1.5 and 2.0 $m^3 \cdot s^{-1}$ from Oct to Dec in all cases – Figure 1.3 in Section 1.3). Changes to the thermal regime have caused large differences in the 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 late April or early May, with a trend to slightly later timing at downstream sites due to the cooling of water as it flows downstream in the fall months when air temperatures are falling (Figure 3.3).

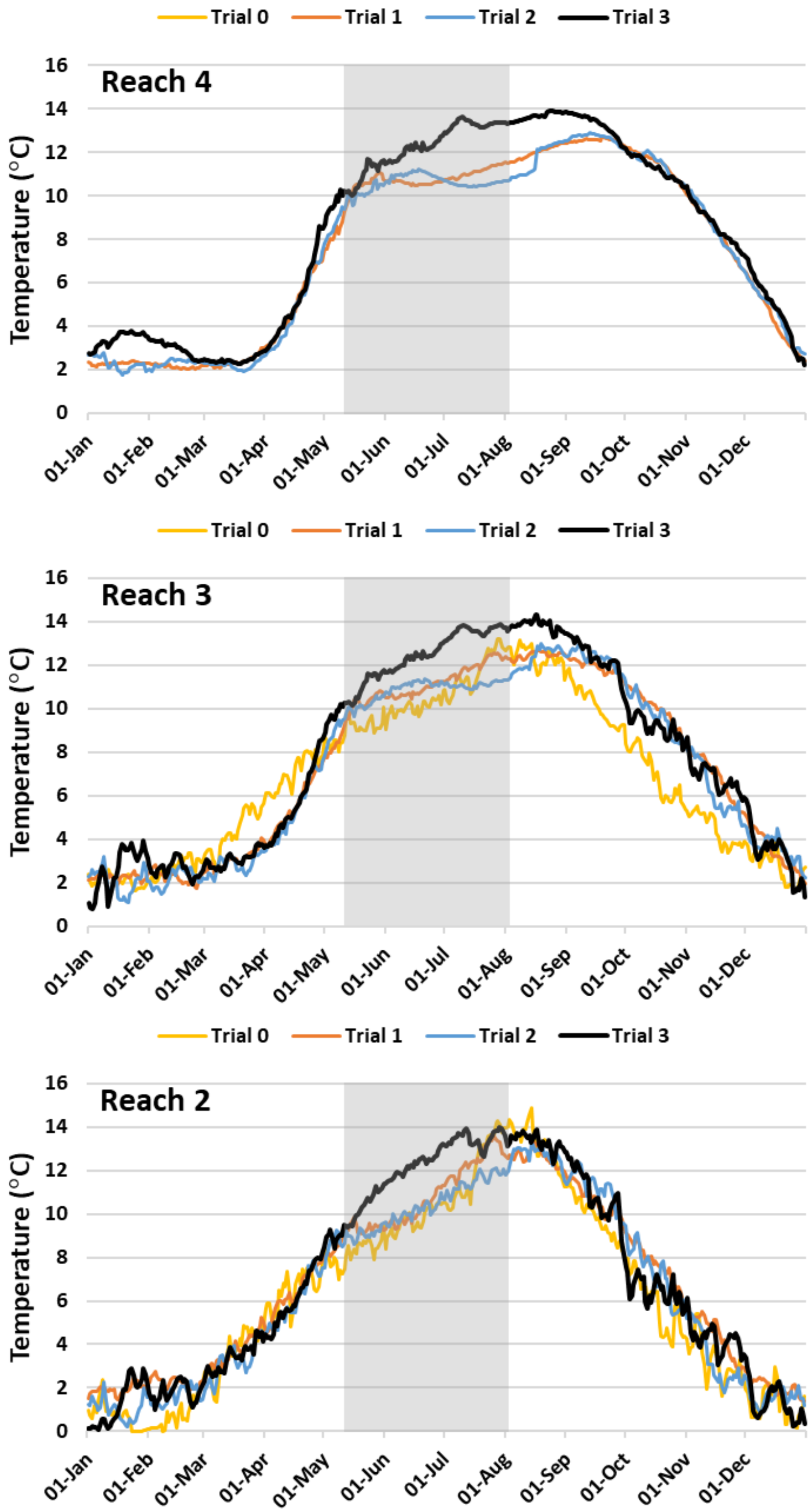


Figure 3.2 Mean daily water temperatures during Trial 0 (pre-flow), Trial 1 (3 $m^3 \cdot s^{-1}$), Trial 2 (6 $m^3 \cdot s^{-1}$), and Trial 3 (high flows) for Reach 4 (top), Reach 3 (middle) and Reach 2 (bottom). The shaded area in each plot depicts the typical Trial 3 high flow period.

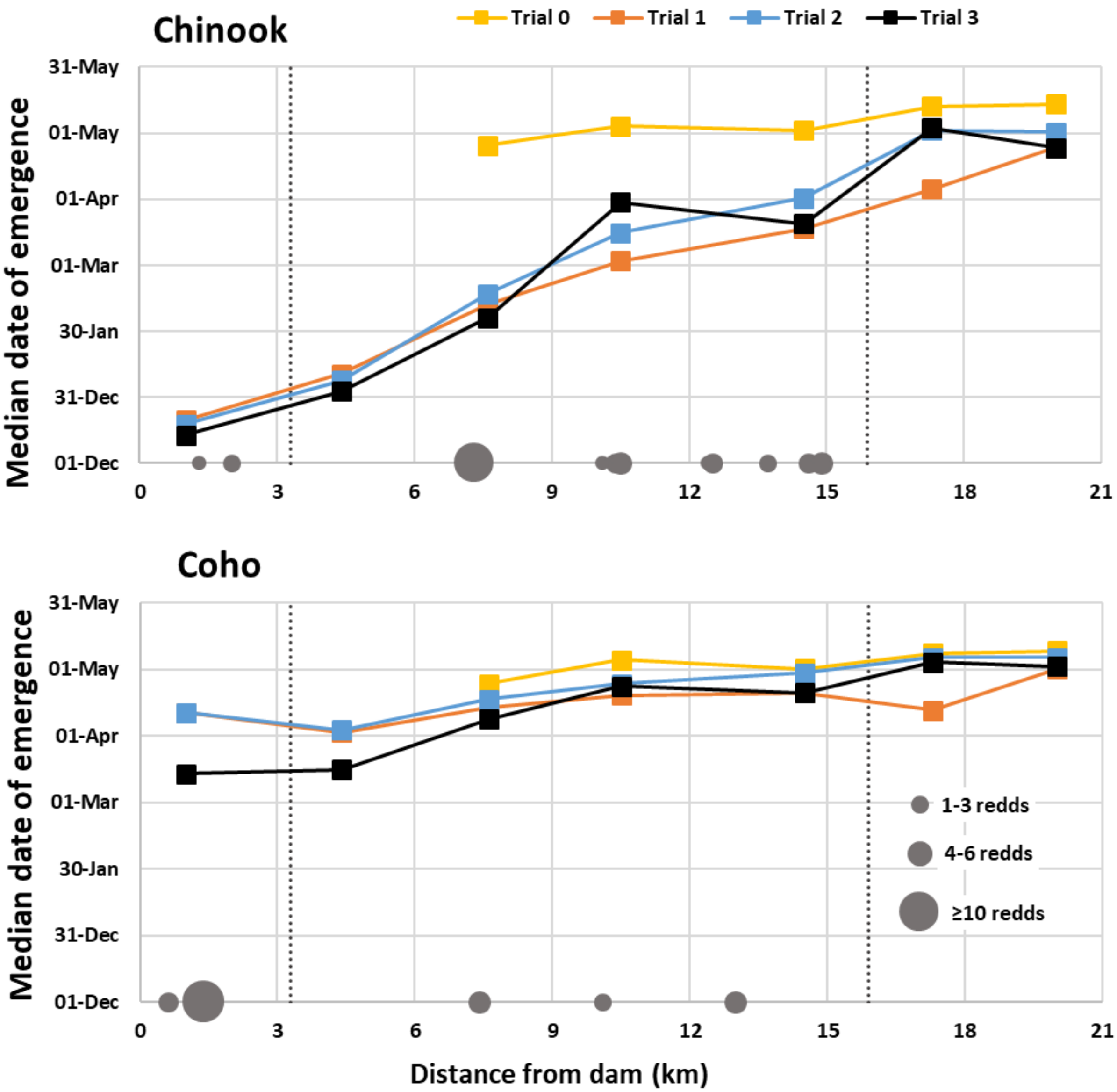


Figure 3.3 Predicted median emergence dates for chinook (top) and coho (bottom) salmon fry at varying distances below Terzaghi Dam based on observed average temperatures for each flow treatment. The vertical dashed lines indicate the location of reach breaks. The locations of redds observed for each species during the Trial 3 years (2016 - 2018) are represented by the grey dots along the x-axis on each plot (data provided by BRGMON-3). The size of dot reflects the relative number of redds at each location. Note: redd data for Reach 2 were not available for this report.

After the initiation of flow from the dam in Trial 1, predicted emergence time for chinook salmon advanced by 1-4 months with the greatest change occurring at sites nearest the dam. The 0.5 $m^3 \cdot s^{-1}$ reduction in October-January flows under Trial 2 and 3 compared to Trial 1 (Figure 1.2) resulted in no discernible change at the first 3 stations below the dam (i.e., Rkm

39.9, 36.5 and 33.3), but effected a slight delay in predicted emergence timing at the other stations (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 3.3). However, release temperatures during the Trial 3 high flow years tended to be warmer than the previous flow trials during the January and February period (see Reach 4 plot in Figure 3.2). The reason for the warmer temperatures during this period are not clear, but shifted the predicted median emergence dates for coho fry between 18 and 28 days earlier, from mid-April to mid-March, in Reach 4 and the top of Reach 3. It is unclear what effect, if any, this potential shift in emergence timing may have had for coho recruitment in Trial 3, but may have contributed to the larger mean size for this species during the fall stock assessment relative to the previous flow trials.

Chinook and coho spawners have utilized spawning areas in both reaches 3 and 4 during Trial 3, but the distribution of redds among those reaches has been different for the two species (Table 3.2 and Figure 3.3). The total number of chinook redds observed from 2016 to 2018 was 42. Eight percent of those were observed in Reach 4, and the remaining 92% were distributed across Reach 3. Based on these findings, approx. 8% of the spawned eggs would be associated with a predicted median emergence (PME) timing of mid-December (near temperature monitoring site A), and a further 38% would have a PME timing of early February (near site C). The remaining 55% would have had a PME of mid- to late-March in the bottom portion of Reach 3.

Observations for coho redd locations were available for 2018 only, and the total number of coho redds observed in that year was 25. Unlike chinook, a much higher proportion of coho spawning was observed in Reach 4 (58%), with an associated PME timing of mid-March under Trial 3 conditions. The remaining 42% of redds were observed in Reach 3 with corresponding PME timing between 9 and 24 April, according to location.

Table 3.2 Proportion of chinook and coho spawning, according to observed redd locations, by distance from dam and predicted median emergence timing in reaches 3 and 4 for Trial 3 years, 2016 to 2018.

Species	Reach	Station (Rkm)	Dist. From Dam (km)	Predicted Median Emergence Date	Percentage of observed redds [a,b]
Chinook	4	A (39.9)	1.0	14-Dec	8% (8%)
	3	B (36.5)	4.4	3-Jan	0% (8%)
		C (33.3)	7.6	6-Feb	38% (45%)
		D (30.4)	10.5	31-Mar	23% (68%)
		E (26.4)	14.5	21-Mar	33% (100%)
Coho	4	A (39.9)	1.0	15-Mar	58% (58%)
	3	B (36.5)	4.4	17-Mar	0% (58%)
		C (33.3)	7.6	9-Apr	17% (75%)
		D (30.4)	10.5	24-Apr	8% (83%)
		E (26.4)	14.5	21-Apr	17% (100%)

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

[b] Values for coho are based on data collected in 2018 only, as this was the only year of redd count data available for this species.

3.1.2. Periphyton

Periphyton algae in the fall was comprised mostly of diatoms with low numbers of Cyanobacteria (commonly known as blue green algae) and rare counts of other taxa (Figure 3.4). In Trials 0 and 1, the common diatom genera included *Achnanthes, Amphipleura, Cocconeis, Cymbella, Cyclotella, Diatoma, Eunotia, Fragilaria, Frustulia, Gomphonema, Hannaea, Melosira, Meridion, Navicula, Nitzschia, Pinnularia, Rhoicosphenia, Rhopalodia, Synedra*, and *Tabellaria.* All of these taxa are commonly encountered in cool mountain streams without pollution. The cyanobacteria during trials 0 and 1 included *Merismopedia* sp., *Oscillatoria* sp., and *Anabaena* sp. In Trial 2, most of the common diatoms found in trials 0 and 1 were found as well as *Campylodiscus* sp, *Didymosphenia* sp., *Gyrosigma* sp., Rossithidium sp., *Stauroneis* sp., and *Staurosira* sp. The cyanophytes at that time included the same genera found earlier and *Arthrospira* sp., *Aphanizomenon* sp., *Gleocapsa* sp., *Lyngbya* 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.* The cyanophytes present in Trial 3 were *Oscillatoria* sp., *Merismopedia* sp., *Pseudanabaena* sp., and *Anabaena* sp.

Among trials 1 and 2, chryso-cryptophytes (*Chroomonas* sp., *Cryptomonas* sp., *Chilomonas* sp., and *Dinobryon* sp.) were found on the periphyton plates. These taxa are flagellated unicells not common to streams. 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 following entrainment in water passing to the Lower Bridge River. They were not found on periphyton plates during Trial 0 when no water was released from Carpenter Reservoir and during Trial 3 when most water was released to the river.

Cell counts increased over the Trial time series in all reaches (Figure 3.4). In Reach 2 mean cell counts changed from 7914 cells x $10^6 \cdot m^{-2}$ in Trial 0 to 55,836 cells x $10^6 \cdot m^{-2}$ in Trial 3. The same pattern of about a seven-fold increase in cell density occurred in Reach 3 between Trials 0 and 3 and in Reach 4 between Trials 1 and 3. The changes were mainly due to the diatoms with a small contribution from the cyanobacteria in all reaches. Figure 3.4 appears to show a large increase in the cyanobacteria but the Y axis is on a log scale to show detail in the low-density values, which is visually deceiving. The actual contribution by the cyanobacteria was small. In Trial 3 when greatest cell densities were found, the cyanobacteria were present at about 0.2% of the total cell density in Reach 2, 0.7% in Reach 3, and 2% in Reach 4. This increasing proportion of cyanobacteria with decreasing distance from the dam suggests the source of these cyanobacteria was Carpenter Reservoir. Furthermore, the cyanobacteria were rare (Reach 3) or absent (Reach 2) in Trial 0 when no flow was released from Carpenter Reservoir, but they increased in importance after the flow release started in Trial 1. This temporal change is further evidence that the cyanobacteria came from Carpenter Reservoir.

Periphyton in the Yalakom River in 2018 was also comprised of diatoms with trace occurrences of cyanobateria (Figure 3.4). Total cell density was <10,000 cells x $10^6 \cdot m^{-2}$, which was among the lowest densities found in the Lower Bridge River.

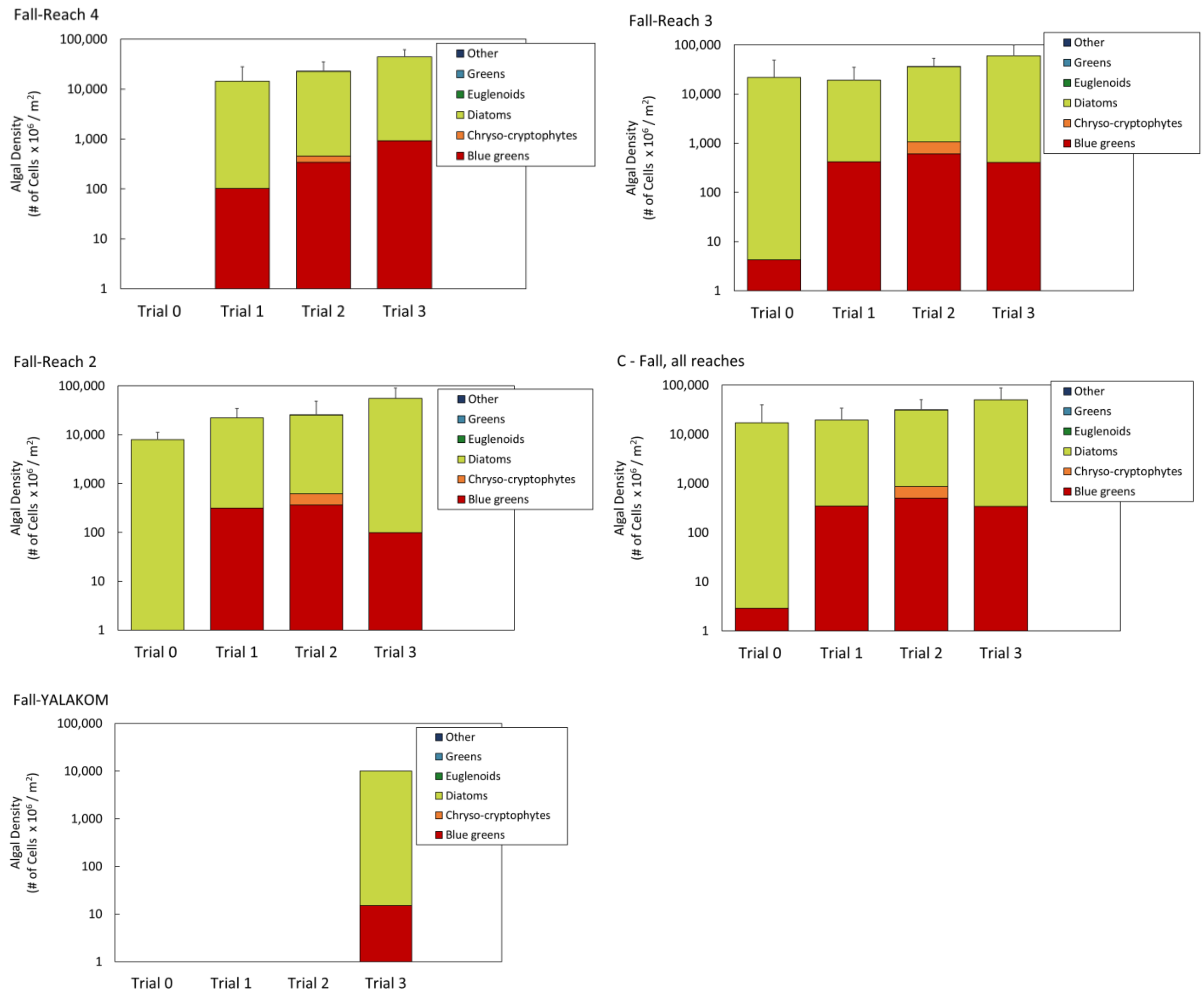


Figure 3.4 Mean algal cell density (± standard deviation) of algal classes in the fall in each of Reach 4 (top left), Reach 3 (top right), Reach 2 (middle left), and all Bridge River reaches (middle right). Yalakom River data collected in 2018 are shown at bottom left.

3.1.3. Benthic Invertebrates

Benthic invertebrates in the Lower Bridge River were from the Ephemeroptera (mayflies), Plecoptera (stoneflies), Tricoptera (caddisflies), Diptera (true flies, including chironomids) and "Other" taxa including Oligochaeta, ostracods, Hemiptera and other true bugs (Figure 3.5).

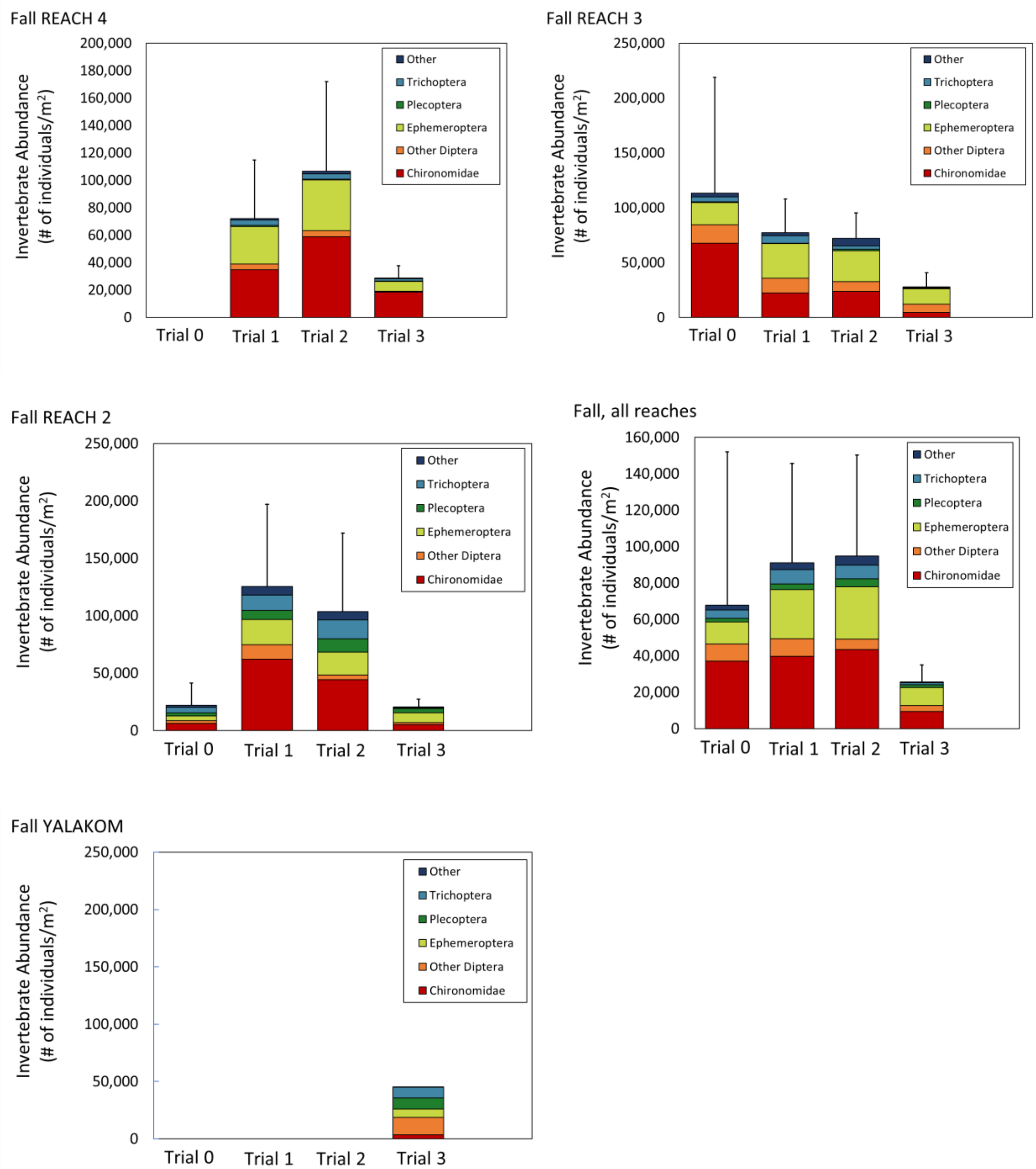


Figure 3.5 Mean abundance (±standard deviation) of invertebrate orders in the fall in each of Reach 4 (top left), Reach 3 (top right), Reach 2 (middle left), and all Bridge River reaches (middle right). Yalakom River data collected in 2018 are shown at bottom left.

Mean density of all taxa combined was 68,000 animals·m^{-2} in Trial 0, it increased by 34% to 91,000 animals·m^{-2} in Trial 1 and to 95,000 animals·m^{-2} in Trial 2 before declining 73% to 26,000 animals·m^{-2} in Trial 3. All taxa contributed to the decline in density between Trials 2 and 3.

Chironomids were most abundant followed by mayflies during Trials 0, 1, and 2. In Trial 3 the chironomids and mayflies occurred in similar densities over all reaches. Dipterans other than the chironomids occurred in greatest densities during Trial 0 and declined thereafter. The caddisfly densities were greatest during Trials 1 and 2 but declined by 86% between Trials 2 and 3 among all reaches. The stonefly density was also greatest during Trials 1 and 2 and it declined by 63% between Trials 2 and 3.

The change in abundances in the fall occurred in different ways among reaches (Figure 3.5). Reach 3 supported relatively high invertebrate densities during Trial 0 (about 5 times that in Reach 2) but with the onset of the 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 highest flows during Trial 3, densities of all taxa declined in all reaches with greatest changes in Reach 4 and Reach 2.

Yalakom River samples included all insect orders found in the Lower Bridge River (Ephemeroptera, Plecoptera, Tricoptera, Diptera) (Figure 3.5). Mean invertebrate density among the 2018 samples was 45,300 animals·m^{-2}, which was less than the mean densities in the Lower Bridge River of 67,700 animals·m^{-2} during Trial 0, 91,000 animals·m^{-2} in Trial 1, and 95,000 animals·m^{-2} in Trial 2. It was greater than the mean density of 25,700 animals·m^{-2} found among all reaches of the Lower Bridge River in Trial 3.

The PERMANOVA revealed a significant interaction between Trial and Reach affecting the benthic invertebrate assemblages ($p=0.019$). This finding showed that changing invertebrate assemblages between reaches was different between the four trials. Effect of the factors, Reach and Trial, were also significant ($p=0.001$ for both). If the interaction was to be used to interpret a “treatment effect” the individual factors could not be examined independently but that situation was not the case here. We were not testing a treatment effect, making it acceptable to examine the distribution of experimental units (samples in a year) by Reach and Trial as long as conclusions were developed from interpreting the interaction.

Variation among invertebrate assemblages by Reach is apparent in Figure 3.6. Polygons drawn around the observations in ordination space show separation by Reach. Distance on the ordination is proportional to difference in assemblage pattern. This visualization shows that Reach 4 assemblages were most dissimilar from those in Reach 2, with Reach 3 assemblages in between. There was more overlap of Reach 2 and Reach 3 assemblages, meaning they were more similar to each other than they were with Reach 4 assemblages.

Dissimilarity by Trial is shown in Figure 3.7. It shows low dissimilarity among assemblages in Trials 1 and 2 because of large overlap of most observations in those two polygons. In contrast, Trial 3 observations were shifted upwards, indicating greater dissimilarity of assemblages in

Trial 3 from those in Trials 1 and 2. Trial 0 observations had a wide distribution in ordination space without clear pattern relative to those in the other Trials.

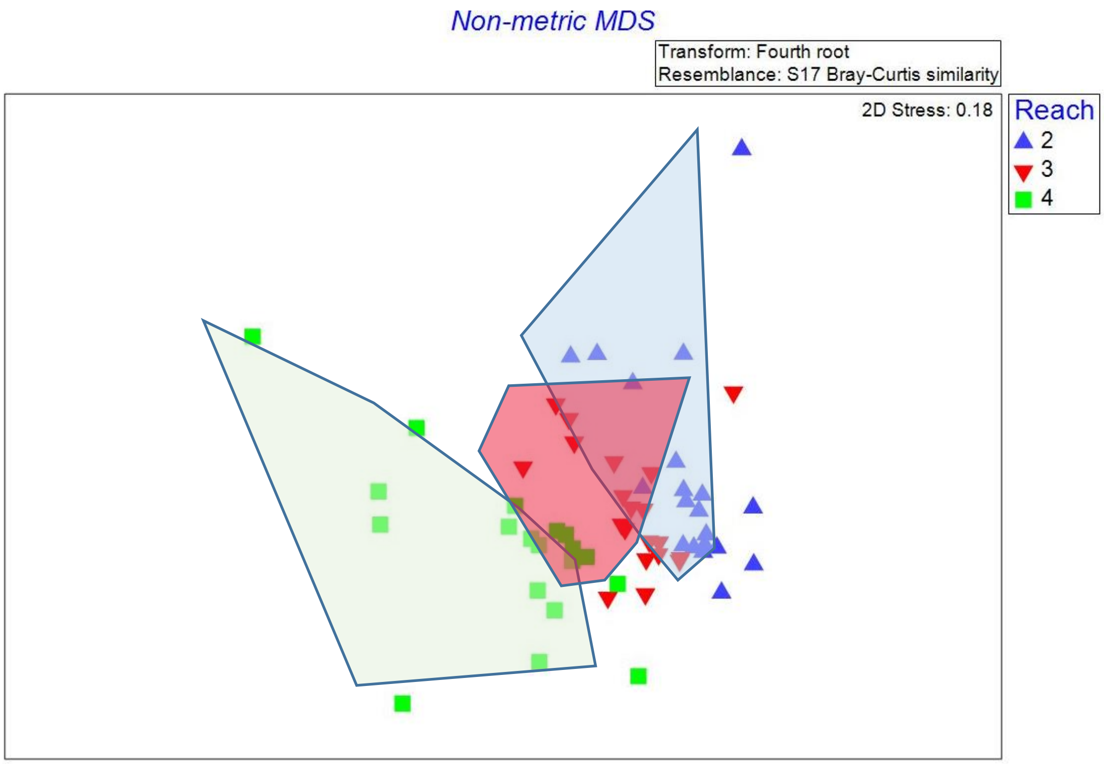


Figure 3.6 Non-metric multidimensional scaling (NMDS) plot contrasting family abundance using Bray-Curtis dissimilarity by Reach among all trials in the fall. Experimental units shown as the coloured symbols were the group of samples collected in a year (Error! Reference source not found.**). Coloured polygons enclose the experimental units by Reach. Note that this ordination looks different from a similar plot produced in 2018 (Sneep et al. 2018) because addition of new data, in this case new data from 2018, changes the distribution of observations in ordination space.**

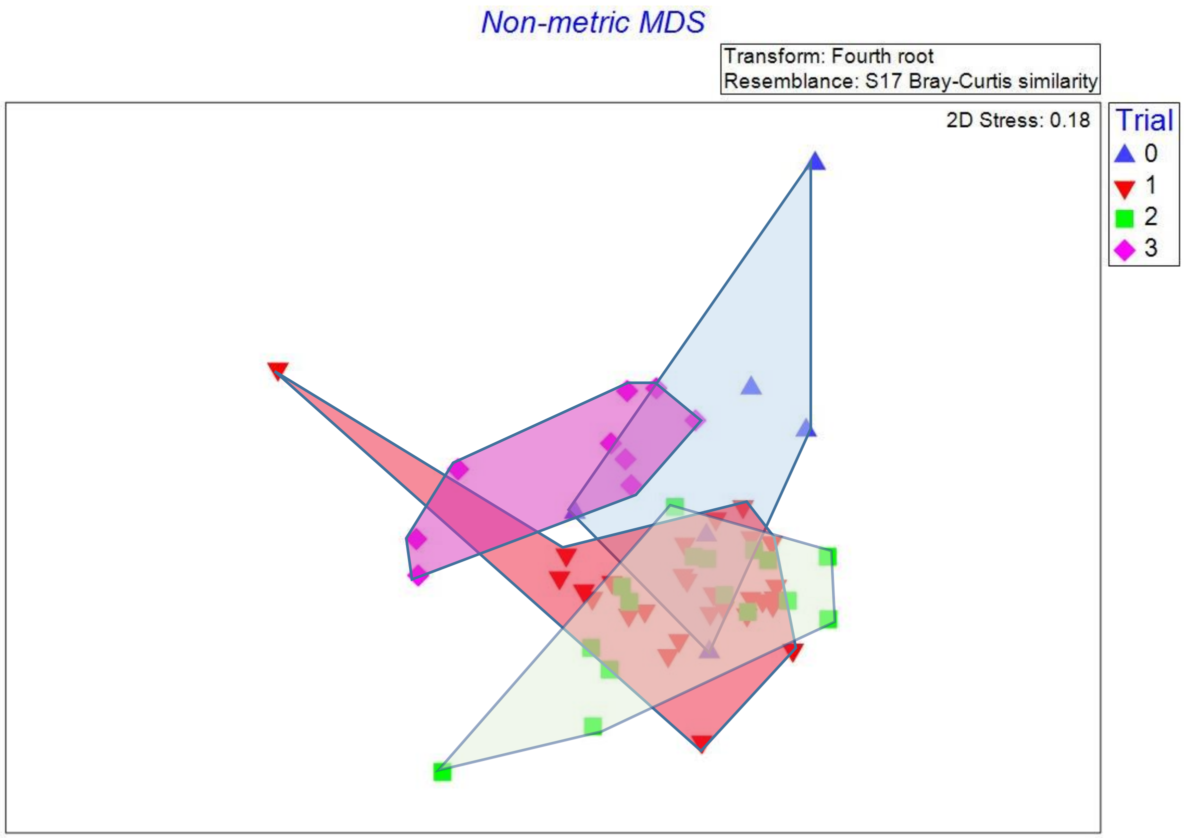


Figure 3.7 Non-metric multidimensional scaling (NMDS) plot contrasting family abundance using Bray-Curtis dissimilarity by Trial among all reaches in the fall. Experimental units shown as the coloured symbols were the group of samples collected in a year (Error! Reference source not found.**). Coloured polygons enclose the experimental units by Trial. Note that this ordination looks different from a similar plot produced in 2018 (Sneep et al. 2018) because addition of new data, in this case new data from 2018, changes the distribution of observations in ordination space.**

SIMPER revealed six taxonomic groups accounting for 70% or more of the dissimilarity among the invertebrate assemblages between Trials (**Table** 3.3) and between Reaches (Table 3.4). Chironomidae (midges) was the most abundant family and it accounted for the highest percent contribution to the dissimilarity between trials (31% to 41%). Simuliidae (black fly larvae), another dipteran family, contributed 9 – 16% of the dissimilarity between Trials. The remaining four families included three mayflies (Baetidae, Heptageniidae, Ephemerillidae), and one caddisfly (Hydropsychidae). Each contributed to 18% or less of the dissimilarity of assemblages between Trials. These same families accounted for most of the dissimilarities among assemblages between Reaches.

Table 3.3 SIMPER output showing invertebrate families contributing to ≥70% of Bray Curtis dissimilarities between Trials in the fall.

Family	Mean abundance (number per m^2 ± SD)				% Contribution to dissimilarity between Trials					
	Trial 0	Trial 1	Trial 2	Trial 3	0 & 1	0 & 2	0 & 3	1 & 2	1 & 3	2 & 3
Chironomidae	37000 ± 74834	39620 ± 39644	43310 ± 36025	9429 ± 8834	44.2	42.7	31.1	40.4	38.2	43.6
Simuliidae	9453 ± 12310	17130 ± 11881	5500 ± 6619	3167 ± 4691	14.5	9.2	16.2	10.4	11.2	
Baetidae	8000 ± 9619	14357 ± 10000	11810 ± 13572	2905 ± 2095	9.1	7.0	10.2	13.0	18.0	11.8
Heptageniidae	1881 ± 1762	7953 ± 5072	7500 ± 5072	4167 ± 2905	7.9	10.8	12.7		7.1	7.0
Ephermerellidae		4405 ± 4286	9167 ± 6667					9.6		9.5
Hydropsychidae	2952 ± 3691		5357 ± 8905			6.5				

Table 3.4 SIMPER output showing invertebrate families contributing to ≥70% of Bray Curtis dissimilarities between Reaches in the fall.

Family	Mean abundance (number per m^2 ± SD)			% Contribution to dissimilarity between Reaches			
	Reach 4	Reach 3	Reach 2	4 & 2	4 & 3	4 & 2	3 & 2
Chironomidae	39882 ± 38287	26905 ± 41048	40691 ± 44334	40.0	36.6	40.0	36.5
Simuliidae	3381 ± 6857	11405 ± 12191	7072 ± 8762	9.3	15.6	9.3	14.3
Baetidae	14429 ± 14810	13810 ± 8595	5738 ± 5810	11.0	14.7	11.0	12.6
Heptageniidae	4786 ± 5167				9.9		
Ephermerellidae	7905 ± 7286		3548 ± 2929	6.9		6.9	
Hydropsychidae	3024 ± 3310	1976 ± 1976	7905 ± 9072	7.3		7.3	6.9

All of these families found in SIMPER to distinguish Trials and Reaches were called invertebrate indicators. To simplify plotting of Trial x Reach interactions of their abundances, the mayflies were rolled up into a single indicator called “mayfly indicator” calculated as the sum of abundances of Baetidae, Heptageniidae, and Ephemerillidae. The caddisfly family, Hydropsychidae, was dropped because it only contributed to dissimilarities in one Trial pair comparison (Trial 0 and 2) and it contributed weakly to the Reach comparisons. These decisions resulted in 3 family indicators (Chironomidae, Simuliidae, and mayfly indicator). Two other common metrics were examined to interpret the interactions: total abundance, and family richness. These common metrics provided an overall perspective on temporal and spatial change in abundance and diversity.

Values of the indicators were plotted by Trial and Reach to show the significant interactions revealed by the PERMANOVA (Figure 3.8). The Chironomidae abundance in Reach 4 increased by about 20% between Trials 1 and 2 and declined by almost the same amount between Trials 2 and 3. In contrast, in Reach 2 they increased almost 10-fold between Trial 0 and 1 but then declined to end up in Trial 3 at about the same density found in Trial 0. A further contrast was in Reach 3 where the Chironomidae declined over all trials by about 30 times. Similar trajectories by the Simulidae over trials by reach added to the significant interactions. The mayfly indicator showed a different pattern. Mayfly density increased between Trials 0 and 1 and then declined in Trial 2 and again in Trial 3 in parallel in both of Reaches 2 and 3. In Reach 4, the mayflies showed a temporal pattern that was similar to that found among the Chironomidae. The Trial x Reach interactions among total invertebrates was very similar to patterns among the indicators, particularly the Chironomidae. This finding is not surprising and it corroborates SIMPER output, as it should, in showing that the indicator families were drivers of Trial x Reach change in the whole benthic invertebrate community.

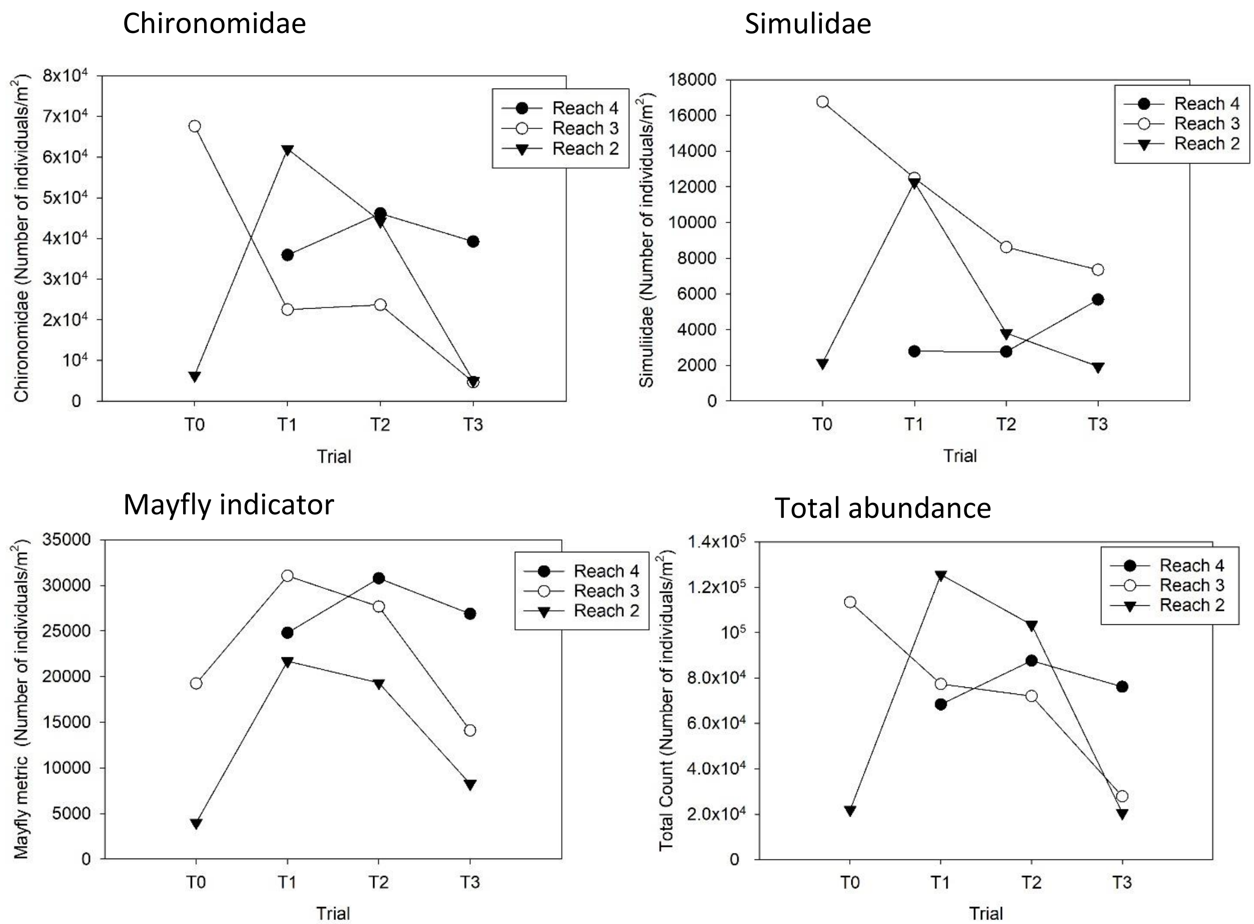


Figure 3.8 **Interactions of mean abundance of the family indicators (Chironomidae (top left), Simuliidae (top right), and mayfly indicator (bottom left)) between Trial and Reach in the fall. The Trial x Reach interaction for total invertebrate abundance is shown for reference at bottom right.**

A plot of family richness is shown for reference in Figure 3.9. In Reach 3, richness of 24 families per sample in Trial 0 steadily declined over trials to end at 18 families per sample in Trial 3. In Reach 2, richness increased from 21 families per sample in Trial 0 to reach 24 families per sample in Trial 2 before declining to 17 families per sample in Trial 3. The overall decline in richness between Trials 2 and 3 shows that conditions favoured fewer families in Trial 3 compared to Trial 2.

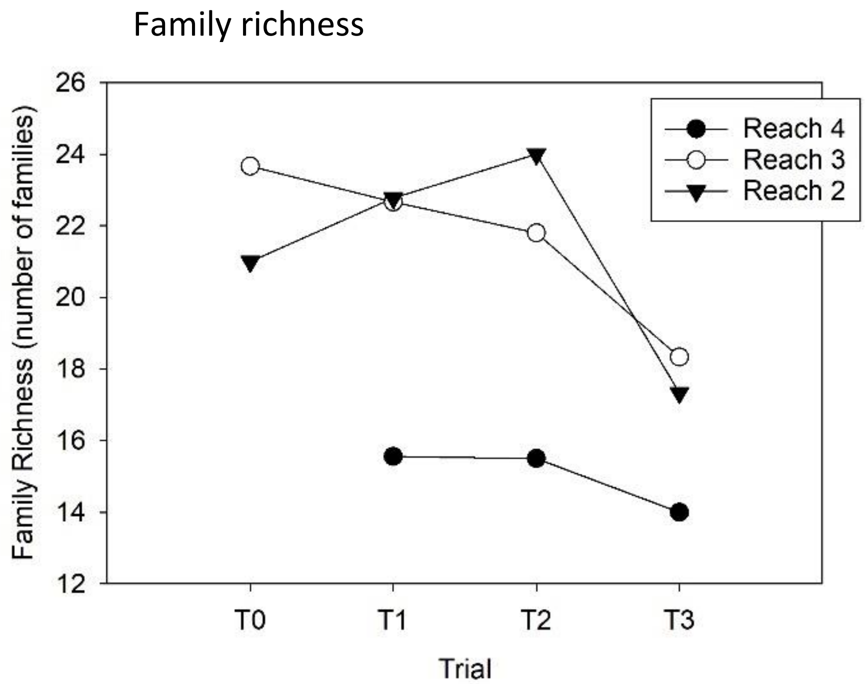


Figure 3.9 Interaction of family richness between Trial and Reach in the fall.

A total of 13 habitat variables was selected using the ecological criteria for RDA shown in Table 2.6. A correlation matrix showed several pairs of these variables where highly co-linear, which would make the RDA unstable (more than one variable explaining the same variance in assemblage patterns). All habitat variables including the ones that were correlated are shown in Table 3.5.

Table 3.5 Correlation matrix of habitat variables considered for inclusion in the benthic invertebrate redundancy analysis. Yellow shading shows pairs of variables that were highly co-linear, meaning that one variable of each co-linear pair had to be removed before running the RDA.

	pink	din	tp	temp	dist_from_origin	dist_from_dam	Incubflow	yalakom	Disturbflow	Peakdisturbflow	PeakPB	sample_depth	sample_velocity
pink													
din	0.48												
tp	0.48	0.72											
temp	0.02	-0.34	-0.02										
dist_from_origin	0.01	0.34	0.08	-0.78									
dist_from_dam	0.02	0.39	0.10	-0.83	0.95								
Incubflow	0.04	0.32	0.14	-0.56	0.89	0.82							
yalakom	0.01	0.32	0.07	-0.76	1.00	0.92	0.89						
Disturbflow	-0.08	0.00	-0.23	-0.40	0.28	0.25	0.23	0.28					
Peakdisturbflow	-0.01	-0.02	-0.23	-0.38	0.28	0.26	0.23	0.28	0.97				
PeakPB	0.23	0.18	0.22	-0.01	-0.27	-0.26	-0.24	-0.27	0.09	0.04			
sample_depth	-0.03	0.08	0.07	0.00	-0.11	-0.09	-0.03	-0.12	0.01	-0.02	-0.03		
sample_velocity	0.16	0.40	0.31	-0.13	0.33	0.36	0.34	0.32	-0.44	-0.43	-0.20	-0.14	

The first co-linear pair was DIN and TDP (r=0.72). These nutrient concentrations varied much the same way over space and time so one of them had to be removed. The best one to select would be the one that most strongly limited biological production. The molar ratio of bio-available N:bio-available P in water can indicate the relative supply of N and P for biological production in streams. In this ratio, bio-available N can be approximated as the sum of NH_4-N and NO_3-N concentration (DIN) and bio-available P can be approximated as soluble P (TDP) concentration. Rhee (1978) showed that for a given species of algae, there is a sharp transition between P-limited and N-limited growth. The particular N:P ratio at which the transition between N and P-limitation occurs is species dependent, varying from as low as 7:1 for some diatoms (Rhee and Gotham 1980) to as high as 45:1 for some blue-greens (Healey 1985). Among periphyton communities that include many algal species, the growth of most will be N-limited at low ratios and P-limited at high ratios. Below a molar N:P of 20, the growth of most algal species will be limited by N whereas P-deficient growth is prevalent at molar N:P ratios greater than 50 (Guildford and Hecky 2000). Because an optimum N:P ratio (above which P limitation occurs and below which N limitation occurs) can vary widely among freshwater algae, the range between 20 and 50 may be regarded as a transition range in a community where some species will be P-limited and others will be N-limited. The Lower Bridge River had molar N:P ratios mostly showing potential N-deficiency (Table 3.6). In Trials 1 to 3, the N:P increased with distance from the dam, indicating increased loading of N, increasing demand for P, or both upstream to downstream. With the exception of Reach 2 during Trial 3, the N:P values showed mostly N deficiency. This rationale supported selection of DIN rather than TDP in the RDA.

Table 3.6 Mean molar N:P by Trial and Reach in the Lower Bridge River.

Reach number	Mean Molar N:P ± SD			
	Trial 0	Trial 1	Trial 2	Trial 3
Reach 4	No water	6 ± 3	17 ± 10	20 ± 6
Reach 3	15 ± 10	10 ± 3	24 ± 11	27 ± 13
Reach 2	11 ± 3	16 ± 4	40 ± 23	62 ± 21

The next co-linear variables included several pairs. Water temperature was strongly correlated with all variables describing distance in the river and influence of the Yalakom inflow (distance from origin, distance from the dam, Yalakom inflow influence: $r \geq 0.76$). These correlations showed that temperature changed with place in the Lower Bridge River. Given that temperature is well known to affect invertebrate growth, it was selected over the more arbitrary distance metrics to include in the RDA. Potential recruitment inferred by the distance metrics was still a variable that could not be ignored, but it was not well described as something independent of temperature. This outcome resulted in all the distance metrics and the binary coding for influence of the Yalakom inflow to be removed from the RDA.

Incubation flow was strongly correlated with the distance metrics. Given that those distance metrics were removed for reasons in the above paragraph, incubation flow remained in the RDA.

Finally, peak disturbance flow and disturbance flow were co-linear. We considered peak flow to best describe the flow Trials because it was so different between Trials. For this reason, disturbance flow was removed and peak flow remained in the RDA.

Eight habitat variables remained in the RDA (

Table 3.7). DIN concentrations increased upstream to downstream in Trials 1 to 3. In Trial 0 the DIN concentrations were more variable, introducing uncertainty about change in concentration with place in the river. Water temperature decreased upstream to downstream. This effect was most pronounced in Trials 1 to 3 when water was released from the dam. Fall is a time of mixing very warm surface water with cool bottom water in Carpenter Reservoir following stratification in the summer (Limnotek 2018). The result was warm water released from the low-level outlet at the dam to the Bridge River in the fall. Due to cool air temperatures in the fall, that water cooled as it flowed downstream to the confluence with the Yalakom River. The Yalakom was cooler still, which resulted in another drop in water temperature at full mixing downstream of the Yalakom in Reach 2.

Water depths and velocities at the samplers were 0.17 – 0.32 m and 0.05 – 0.42 $m \cdot s^{-1}$ respectively. These conditions were defined somewhat by flow but more by where the samplers were placed during each incubation. They were not placed in deep fast water that could compromise worker safety or risk sampler loss. Inadvertently, this placement provided a way to reduce velocity and depth variance. Incubation flows were those occurring at the sampling site during the sampler incubation periods. They ranged between zero flow in Trial 0 in Reach 4 to 6.3 $m^3 \cdot s^{-1}$ during Trial 3 in Reach 2. Incubation flows were those happening after the bell-shaped hydrograph associated with the Trial flow releases. Peak disturbance flow was directly associated with Trial. It was zero during Trial 0 in Reach 4 and it reached an average of 130 $m^3 \cdot s^{-1}$ during Trial 3 in Reach 2.

Data for these same measurements in the Yalakom River is shown for comparison between the two rivers in

Table 3.7. The Yalakom River was cooler than the Lower Bridge River in the fall and would have contributed to cooling of Reach 2 in the Lower Bridge River among all trials. Mean 2018 flow in the Yalakom River during the time of peak disturbance flow in the Lower Bridge River (May 1 – August 31) was greater than in Reach 3 of the Lower Bridge River during Trials 0 – 2 but about 4 times lower than in that reach during Trial 3. DIN concentration of 81 $\mu g \cdot L^{-1}$ was greater in the Yalakom River than in all places of the Lower Bridge River. PB in the Yalakom was lower than at any time and place in the Lower Bridge River. Water depths and velocities at the periphyton plates and invertebrate baskets in the Yalakom were similar to those in the Lower Bridge River at the same time. Incubation flow in the Yalakom (3.8 $m^3 \cdot s^{-1}$) was greater than it was immediately upstream in Reach 3 of the Lower Bridge River (2.0 $m^3 \cdot s^{-1}$).

Table 3.7 Mean values (± standard deviation) of measured habitat attributes in the Lower Bridge River in the fall selected for the benthic invertebrate RDA. Yalakom River data are shown for comparison.

Trial	Reach	DIN ($\mu g \cdot L^{-1}$)	Temperature (°C)	Water depth at sampler (m)	Water velocity at sampler ($m \cdot s^{-1}$)	Incubation flow ($m^3 \cdot s^{-1}$)	Peak disturbance flow ($m^3 \cdot s^{-1}$)	Mean PB (μg chl-a $\cdot cm^{-2}$)	Pink (on/off)
0	4	No data because the reach was dewatered							
	3	45 ± 27	8.3 ± 2	0.17 ±	0.27	0.3 ± 0.2	10 ± 12	4.3 ± 2.6	On/off
	2	32 ± 7	7.5 ± 2	0.17	0.41	4.9 ± 1.6	36 ± 14	3.2 ± 2.6	On/off
1	4	24 ± 8	10.8 ± 0.5	0.27	0.26	2.5 ± 0.2	5 ± 0.4	7.6 ± 4	On/off
	3	59 ± 60	9.6 ± 0.9	0.3	0.34	2.9 ± 0.2	6 ± 1	7.6 ± 4.5	On/off
	2	98 ± 80	7.2 ± 0.8	0.26	0.35	6.2 ± 0.6	26 ± 10	5.9 ± 4	On/off
2	4	29 ± 15	10.1 ± 0.4	0.29	0.26	1.5 ± 0.5	17 ± 2	9.0 ± 6.3	On/off
	3	127 ± 130	8.2 ±1	0.32	0.2	1.9 ± 0.5	19 ± 2	12.3 ± 7.9	On/off
	2	151 ± 124	5.6 ± 0.8	0.3	0.42	4.9 ± 0.8	45 ± 7	9.5 ± 6.3	On/off
3	4	16 ± 2	10.2 ± 0.6	0.2	0.05	1.6 ± 0.1	109 ± 14	10.5 ± 4.7	On/off
	3	27 ± 8	8.3 ± 1	0.31	0.17	2.0 ± 0.2	111 ± 13	9.7 ± 5	On/off
	2	55 ± 3	5.8 ± 0.6	0.27	0.15	6.3 ± 0.4	130 ± 11	5.7 ± 3.1	On/off
Yalakom River		81	4.1	0.32	0.2	3.8	27[a]	1.9 ± 0.3	Off

[a] Mean flow in the Yalakom River measured at the same time as peak disturbance flow in the Lower Bridge River (March 1 – August 31, 2018).

All eight habitat variables constrained the ordination in the redundancy analysis (Figure 3.10). The variables were Peak disturbance flow, (peakdisturbflow), Incubation flow (Incubflow), water depth at the sampler (sampler depth), DIN concentration (DIN, hidden behind sampler depth in Figure 3.10), water velocity at the sampler (sampler velocity), water temperature (temp), peak periphyton biomass (PeakPB), and pink salmon on/off (pink). The first two RDA axes explained 71% of fitted variance in assemblage patterns and 34% of total variance among all trials and reaches. Eight axes were calculated with axes 3 to 8 explaining 29% of the fitted variance and only 14% of total variance. Axes 3 to 8 are not shown because of this small addition to explaining variance. Peak disturbance flow and water temperature and to a small extent PeakPB were drivers of RDA axis 2 (the vertical axis) while incubation flow, sample depth, sample velocity, DIN concentration, and to a small extent, pink on/off were drivers of axis 1 (the horizontal axis). The invertebrate assemblages laid along two habitat gradients. One was a gradient of temperature and incubation flow during Trials 0, 1, and 2 (Axis 2) modified by sample depth and velocity (axis 1). The second gradient also laid along a temperature and incubation flow gradient with a large shift in assemblage pattern due to peak disturbance flow during Trial 3. We interpret this interaction as meaning that site-specific water temperature and flow at the time of sampling explained the assemblage patterns in all Trials but peak flow during the flow release in Trial 3 changed that underlying temperature and incubation flow effect on the assemblage patterns, producing different assemblages in Trial 3. These patterns occurred among all reaches.

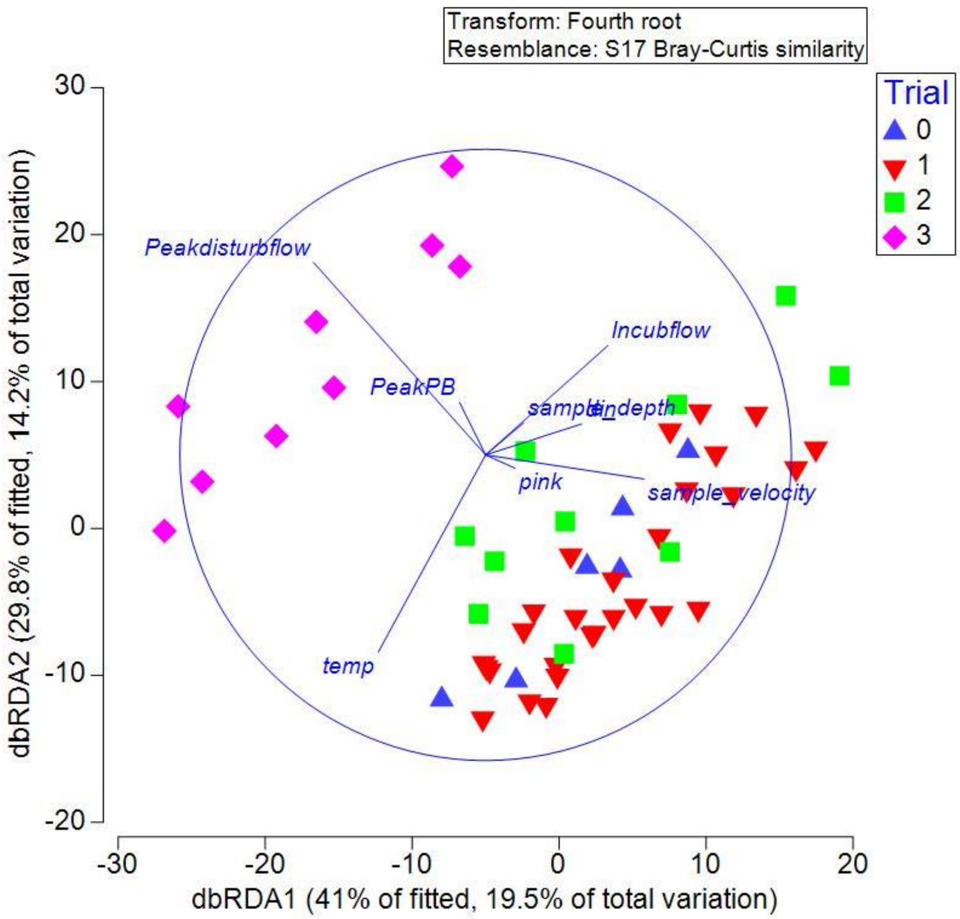


Figure 3.10 Redundancy analysis (RDA) plot of the fall invertebrate family structure as it related to habitat attributes among Trials. Percent of variation explained by each axis is separated as a percent of explained variation by the constraining habitat variables (fitted) and the total variation in the multivariate regression model (total).

3.1.4. Juvenile Fish Production: Size

Mean weight of Age-0+ mykiss in all reaches was almost always higher during the high flow period (2016-2018) compared to other treatment periods (**Error! Reference source not found.**). This likely occurred because of reduced density (see Figure 3.13 and Figure 3.14 in Section 3.1.5). 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 3.5, above) combined with the quality rearing conditions in this reach prior to the flow release. Mean size was also greater during the high flow period for Age-1 mykiss, particularly in reaches 3 and 4; however, there was considerable overlap in standard deviation error bars for Reach 3.

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. 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 3.13 in Section 3.1.5), 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.

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 highest in reaches 3 and 4 during the Trial 3 high flow years, likely due to reduced competition from lower fish densities overall, as well as earlier emergence (relative to Trial 0).

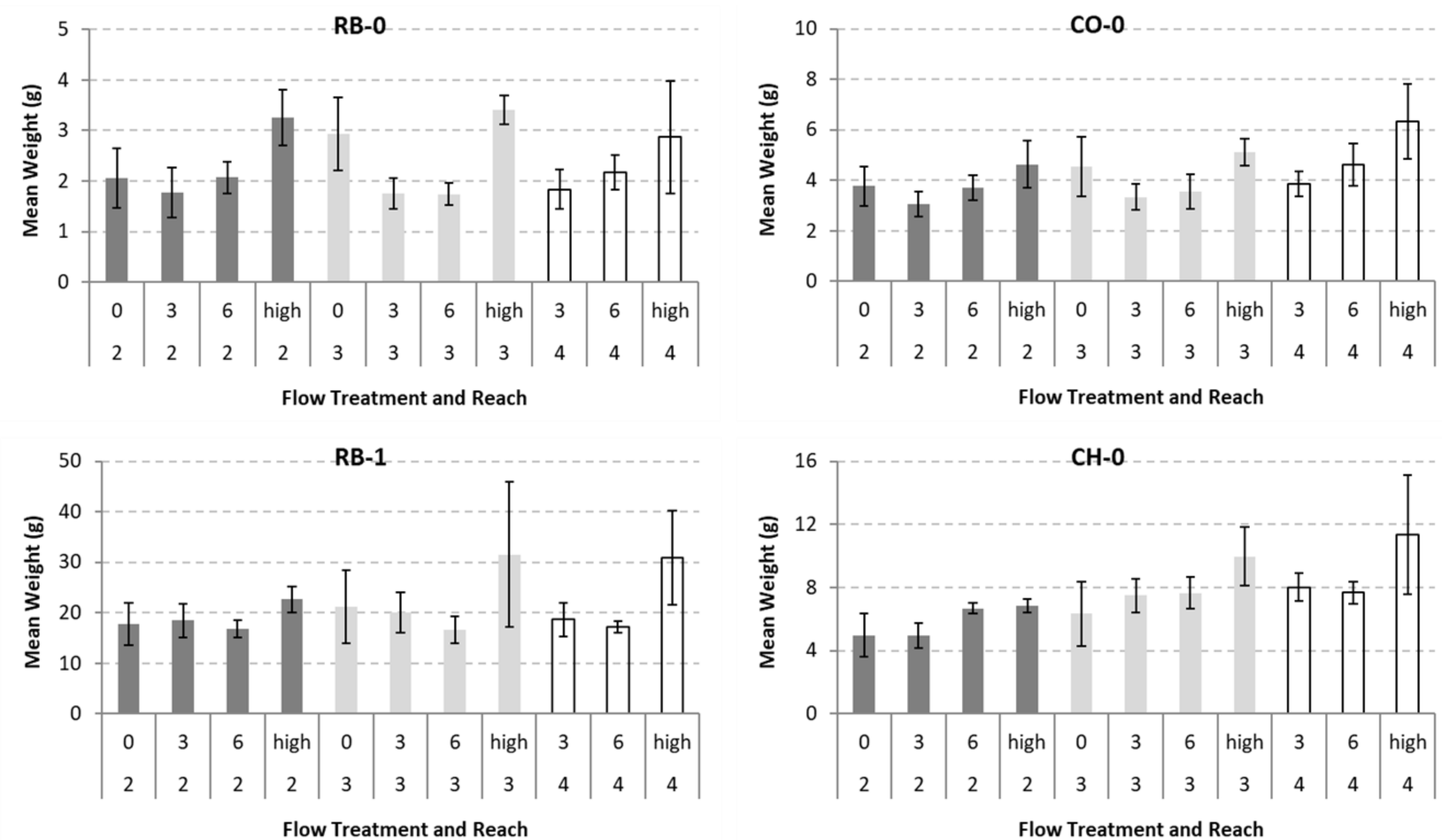


Figure 3.11 Mean juvenile salmonid weight during fall standing stock assessments across flow treatments (0, 3, and 6 $m^3 \cdot s^{-1}$ treatments and the high flow years) and reaches (2, 3, and 4). RB-0, RB-1, CO-0, and CH-0 denote Age-0+ mykiss, Age-1 mykiss, Age-0+ coho, and Age-0+ chinook, respectively. 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).

3.1.5. Juvenile Fish Production: Abundance and Biomass

High Flow Years (2016 – 2018)

Estimated total abundance of juvenile salmonids across the three high flow years (2016 – 2018) was between 60,000 to 80,000 fish across the 3 study reaches, but the contribution of each species and age class varied (**Error! Reference source not found.**). Abundance of mykiss fry was approximately 39,000, 42,000 and 33,000 fish in 2016, 2017 and 2018, respectively. Highest production was in Reach 3, followed by Reach 2, and then Reach 4. Mykiss parr were more abundant in 2016 (19,000 fish) compared to 2017 (11,000 fish) and 2018 (9,000 fish), since this cohort reared for their first full year under the Trial 2 flow regime (in 2015) when mykiss recruitment was higher. Since the production of mykiss fry has been reduced during the high flow years, the corresponding numbers of mykiss parr in 2017 and 2018 were also reduced. Patterns of abundance among reaches were the same as for mykiss fry, but differences among reaches were smaller in 2017 and 2018.

Total coho fry abundance was approximately 10,000, 7,000, and 7,000 fish in 2016, 2017 and 2018, respectively. The numbers in reaches 3 and 2 were fairly equivalent in 2016, but Reach 3 production diminished in 2017 and 2018. Numbers for this species in Reach 4 were low in every high flow year. Chinook fry abundance was lower in 2016 (11,000 fish) than in 2017 and 2018 (14,000 fish in each year). Numbers were consistently higher in Reach 3 than in Reach 2, but only slightly, and production in Reach 4 has remained very low.

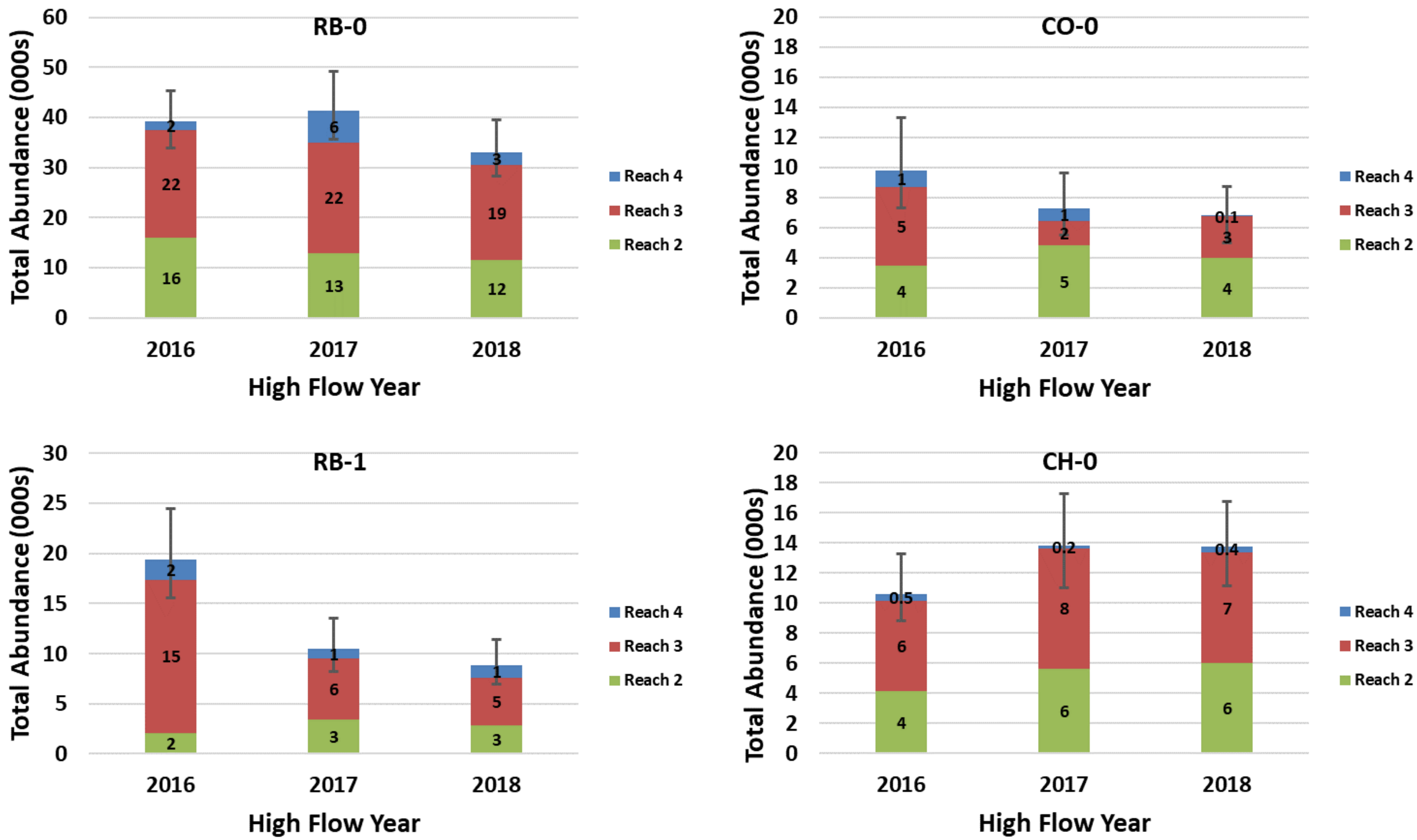


Figure 3.12 **Abundance (in thousands) of age-0 mykiss (RB-0), age-1+ mykiss (RB-1), age-0 coho (CO-0), and age-0 chinook (CH-0) in the Lower Bridge River by reach for each high flow year (2016 – 2018). Vertical lines show 90% credible intervals from posterior distributions of abundance for each year from the hierarchical Bayesian model.**

Comparisons among Flow Trials (0 – 3)

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 3.8 and Table 3.9, Figure 3.13 and Figure 3.14). 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 – 2018) was 0.42-fold of the mean abundance under Trial 0 (i.e., abundance was 58% lower). Relative to Trials 1 and 2, the high flow mykiss abundance was 0.22- and 0.24-fold, or a decline of 78% and 76%, respectively.

Table 3.8 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 $m^3 \cdot s^{-1}$	90	36	25	38
3 $m^3 \cdot s^{-1}$	174	35	81	22
6 $m^3 \cdot s^{-1}$	162	33	76	13
High	38	10	8	13

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	124	326	39	114

Age-1 mykiss abundance increased a small amount in Reach 2 from Trial 0 to Trial 1 while there was a large decrease in Reach 3. Trial 1 produced about 11,000 additional parr in Reach 4. Across reaches there were negligible changes in mykiss parr abundance across the Trial 0, Trial 1, and Trial 2 treatments. 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.

Table 3.9 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)			0.2
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)			0.2

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 3.8a, Figure 3.13 and Figure 3.14). 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 during the high flow years (2016 – 2018) was only 1/3rd of the abundance under Trial 0, and 1/10th of the abundance estimates under trials 1 and 2 (i.e., a 90% reduction).

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 3.8a, Figure 3.13 and Figure 3.14). Chinook recruitment in Reach 4 has been low across all flow treatments. 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, the high flows from 2016 – 2018 have not resulted in a further decline in Age-0+ chinook abundance (relative to Trial 2), perhaps because their abundance was already depressed due to other factors (e.g., water temperatures during the incubation period).

Despite the increased mean weights for virtually every species and age class under the high flows (see **Error! Reference source not found.**, above), differences in biomass among flow treatments closely matched those based on abundance (Table 3.8b, Figure 3.15 and Figure 3.16). This was because the changes in abundance were more substantial than the relative changes in mean weights among treatments. Also, the addition of the 2018 results did not substantively change the high flow estimates (abundance or biomass) for any species and age class since the differences among the high flow years were very small relative to the differences among trials. This is also reflected by the very small 90% credible interval bars for each of the high flow data points in Figure 3.14.

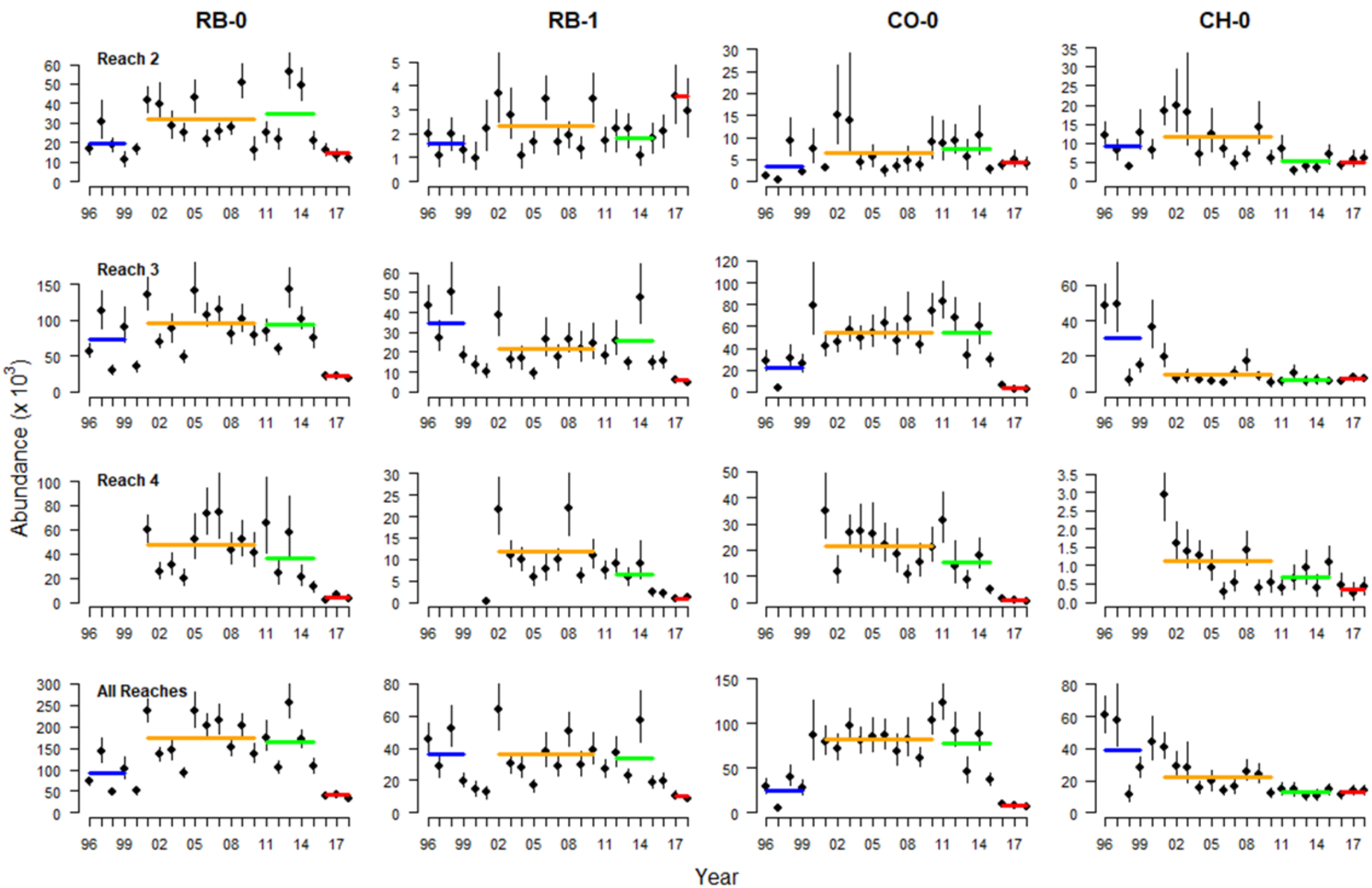


Figure 3.13 Abundance (in thousands) of juvenile salmonids in the lower Bridge River by reach (row) and species-age class (column). Points and vertical lines show mean values and 90% credible intervals from posterior distributions of abundance for each year from the hierarchical Bayesian model, respectively. Blue, orange, green and red lines show the mean values for trials 0, 1, 2, and high flow treatments, respectively. RB-0, RB-1, CO-0, and CH-0 denote age-0 mykiss, age-1+ mykiss, age-0 coho, and age-0 chinook, respectively.

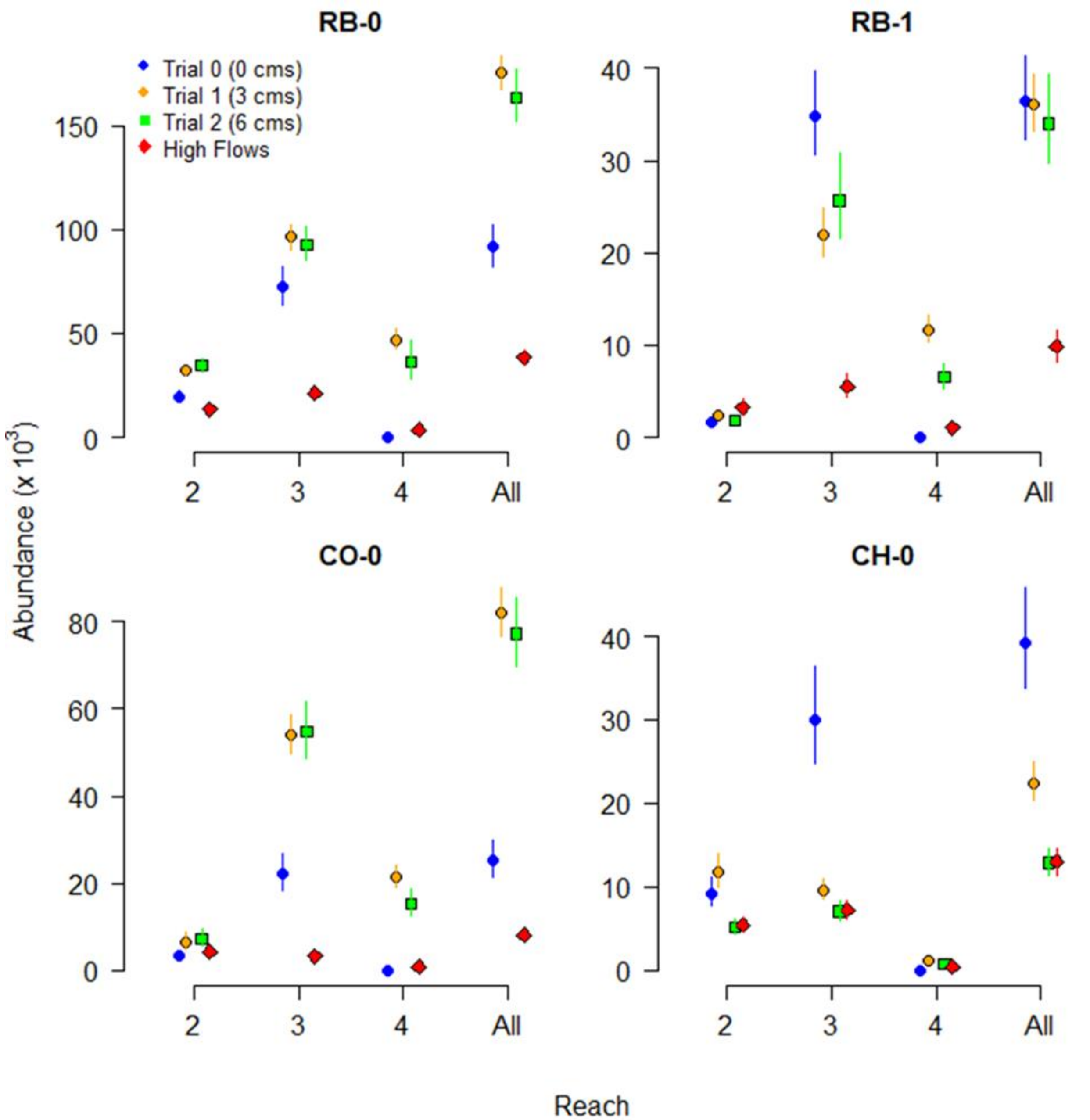


Figure 3.14 Average (points) and 90% credible intervals (vertical lines) of juvenile salmonid abundance by reach for each flow treatment. RB-0, RB-1, CO-0, and CH-0 denote Age-0+ mykiss, Age-1 mykiss, Age-0+ coho, and Age-0+ chinook, respectively.

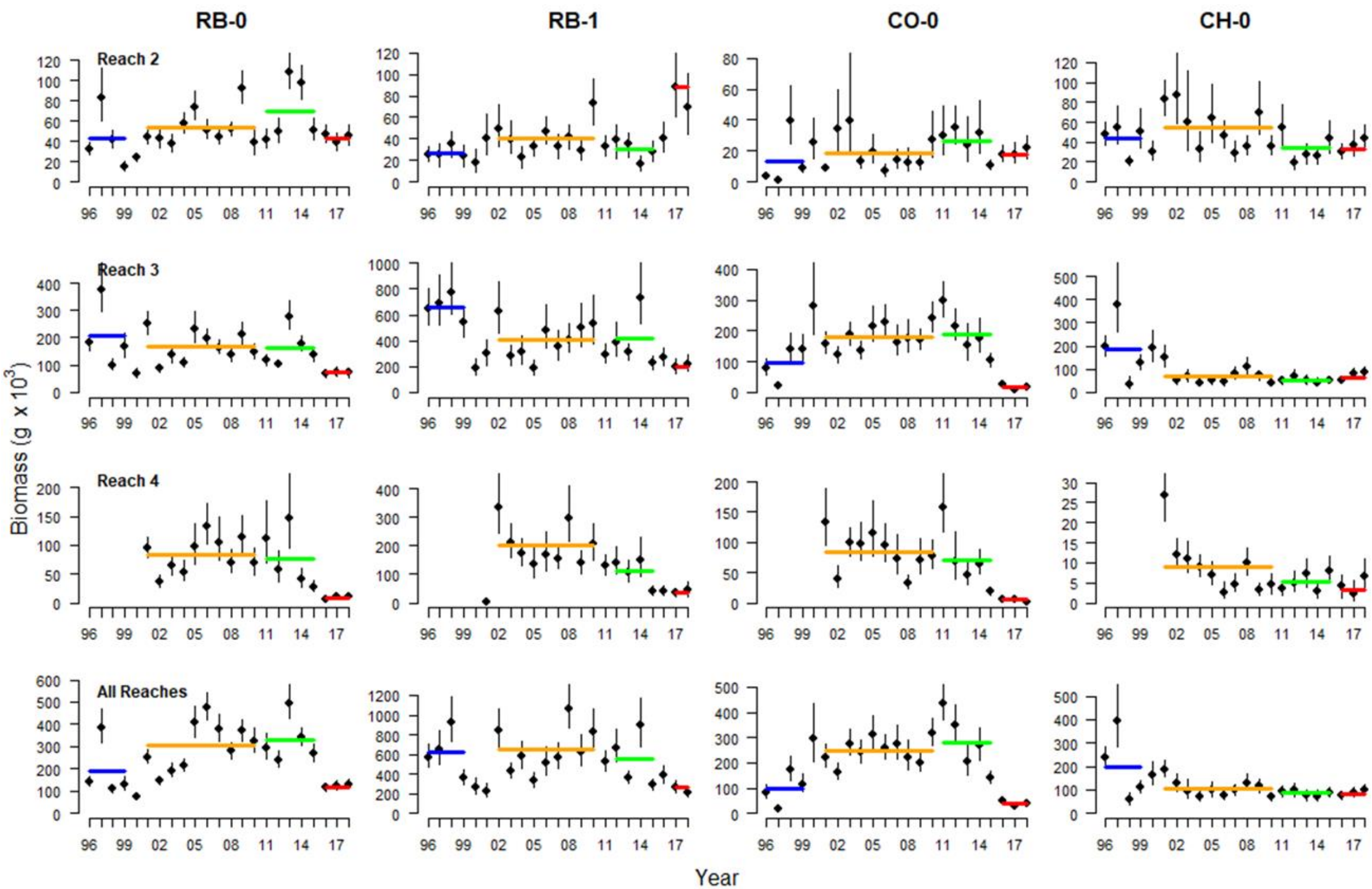


Figure 3.15 Biomass (in thousands of grams or kilograms) of juvenile salmonids in the Lower Bridge River by reach (row) and species-age class (column). See caption for Figure 3.13 for details.

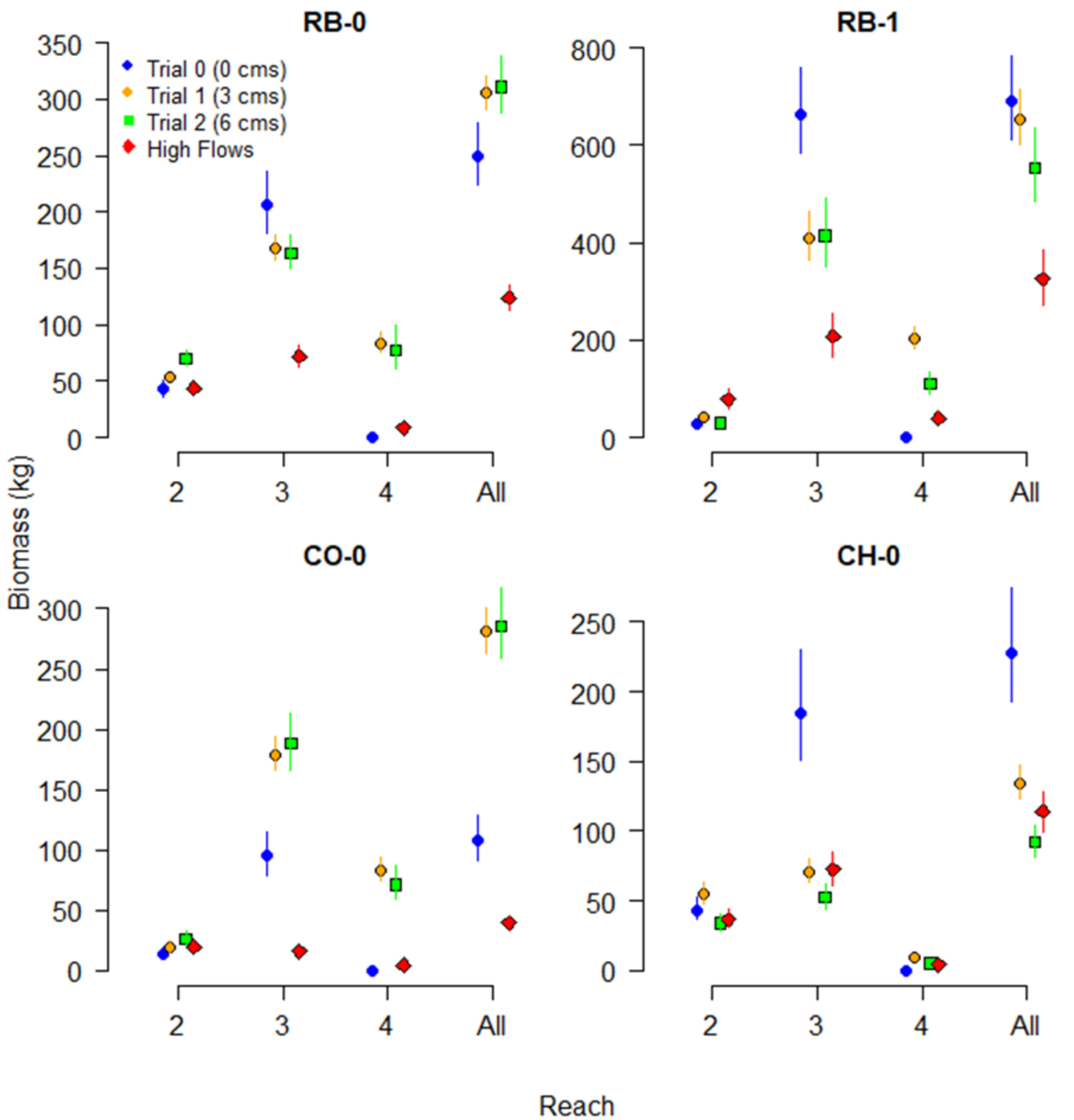


Figure 3.16 Average (points) and 90% credible intervals (vertical lines) of juvenile salmonid biomass by reach for each flow treatment.

Enhanced Off-channel Habitats (2018)

The Bluenose off-channel habitat located in Reach 4 had high densities of Age-0+ mykiss (RB-0), as well as a decent number of mykiss parr (RB-1; Table 3.10). Catches of juvenile coho (CO-0) were very low at Bluenose, and no chinook juveniles (CH-0) were caught. The Applesprings site in Reach 1 had high catches of juvenile coho and small numbers of juvenile chinook and rainbow trout. The Bluenose site had much higher densities of rainbow trout compared to the mainstem in fall of 2018, while the Applesprings site had much higher densities of juvenile coho compared to densities in the mainstem of reach 2 in 2018 (Figure 3.17

top and middle panels). The densities in these enhanced off-channel sites in 2018 were similar to average densities in their associated reaches during trials 1 and 2 (Figure 3.17 bottom panel).

These results demonstrate the use of these two off-channel sites by juvenile salmonids following the period of high flows in 2018, and that they likely function as refuge habitats when rearing conditions in the mainstem are poor. The sum of abundance estimates for juvenile salmonids in 2018 at Bluenose (~1,500) and Applesprings (~5,000) was ~6,500 fish, which was ~10% of the number of fish in the mainstem across reaches 2 – 4 in 2018 (63,000). This was a fair contribution considering the difference in wetted area (i.e., ~1 ha for the off-channel habitats vs. ~50 ha for reaches 2, 3 and 4). However, it is also important to acknowledge that the abundance of mainstem fish has been reduced by 215,000 fish under the high flows (i.e., 284,000 Trial 2 average - 69,000 high flow average). Making up this production without a restoration of flows in the mainstem would require a 33-fold increase in the contribution from off-channel habitats or other restoration works.

Table 3.10 Catch, capture probability (pCap), abundance, density, and biomass of juvenile salmonids at enhanced side channel sites in fall of 2018. Note: the suffix "RI" and "PO" in the site names denotes riffle and pool habitats, respectively.

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

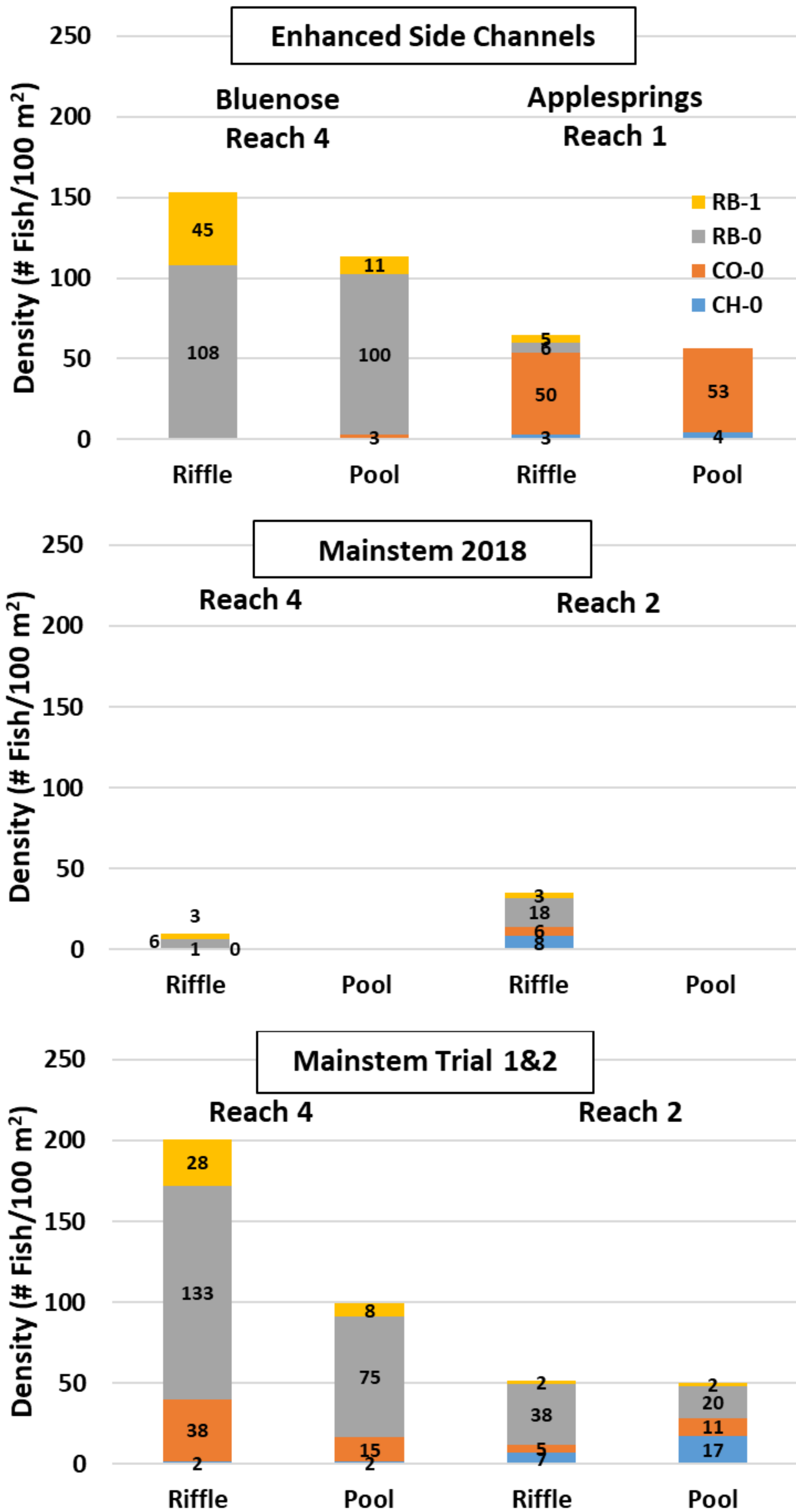


Figure 3.17 **Density of juvenile salmonids in enhanced side channels in fall of 2018 by habitat (top) compared to mainstem densities during the same time period (middle) and average values in the mainstem prior to 2016 (bottom, trials 1 & 2).**

3.1.6. Juvenile Fish Production: Stock-Recruitment

As reported in Year 6 (2017), 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. Due to relative similarities in both the escapement estimates and fry recruitment numbers for both coho and chinook across the 3 high flow years (2016 – 2018), the results of the stock-recruitment analysis did not substantively change relative to what was reported last year.

Age-0+ coho abundance increased under the Trial 1 and Trial 2 treatments relative to the Trial 0 pre-flow period (Figure 3.18). 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 65% under the recent high flows.

There is no indication from the shape of this stock-recruitment curve that coho fry production has been limited by escapement (i.e. under-seeded) as almost all data points are near or on the asymptote of the stock-recruitment curve. For example, escapements seeding the recruitment in 2005, 2009, 2011 and 2015 were in the same range as the 2016 – 2018 estimates (i.e., between 100 – 500 spawners); however, those Trial 1 and 2 years produced between 30,000 to 115,000 more fry than the Trial 3 high flow years. 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 3.19). In this case, almost all of the data points reflect escapements that are less than required to maximize fry production. Based on the data available, this more constrained curve provides a near equivalent fit to the data. The difference in log-likelihood measuring the fit of the curves in Figure 3.18 and Figure 3.19 is less than 2 units and therefore these curves are not significantly different. The stock-recruitment curve in Figure 3.19 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 (as it would be for several Trial 1 and 2 years also). More data are required (i.e., at low escapement levels) to better define the initial slope of the stock-recruitment relationship to strengthen inferences about 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 3.20). 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 (Figure 3.20). Owing to the steep initial slope there is no indication that escapement has been limiting fry abundance, and the 2016 – 2018 datapoints are within a cluster of datapoints from the other flow trials near the asymptote of the curve (e.g., brood years 2008, 2009, 2010, 2011 and 2013). 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 and Trial 3 flow treatments relative to the pre-flow conditions, and no effects of the Trial 2 flow treatment (Figure 3.21). 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 stock-recruit data are not yet ideal for allowing us to clearly differentiate flow effects from potential stock size effects (escapement). Though there is a fairly large sample size overall (*n*= 18 data points for coho and *n*= 20 datapoints for chinook), the data must be parsed according to flow treatment because we have observed different levels of production due to variable incubation and rearing conditions under the different flow trials (Section 3.1.5). As a result, there is a much smaller *n* size for defining the initial slope of each individual curve.

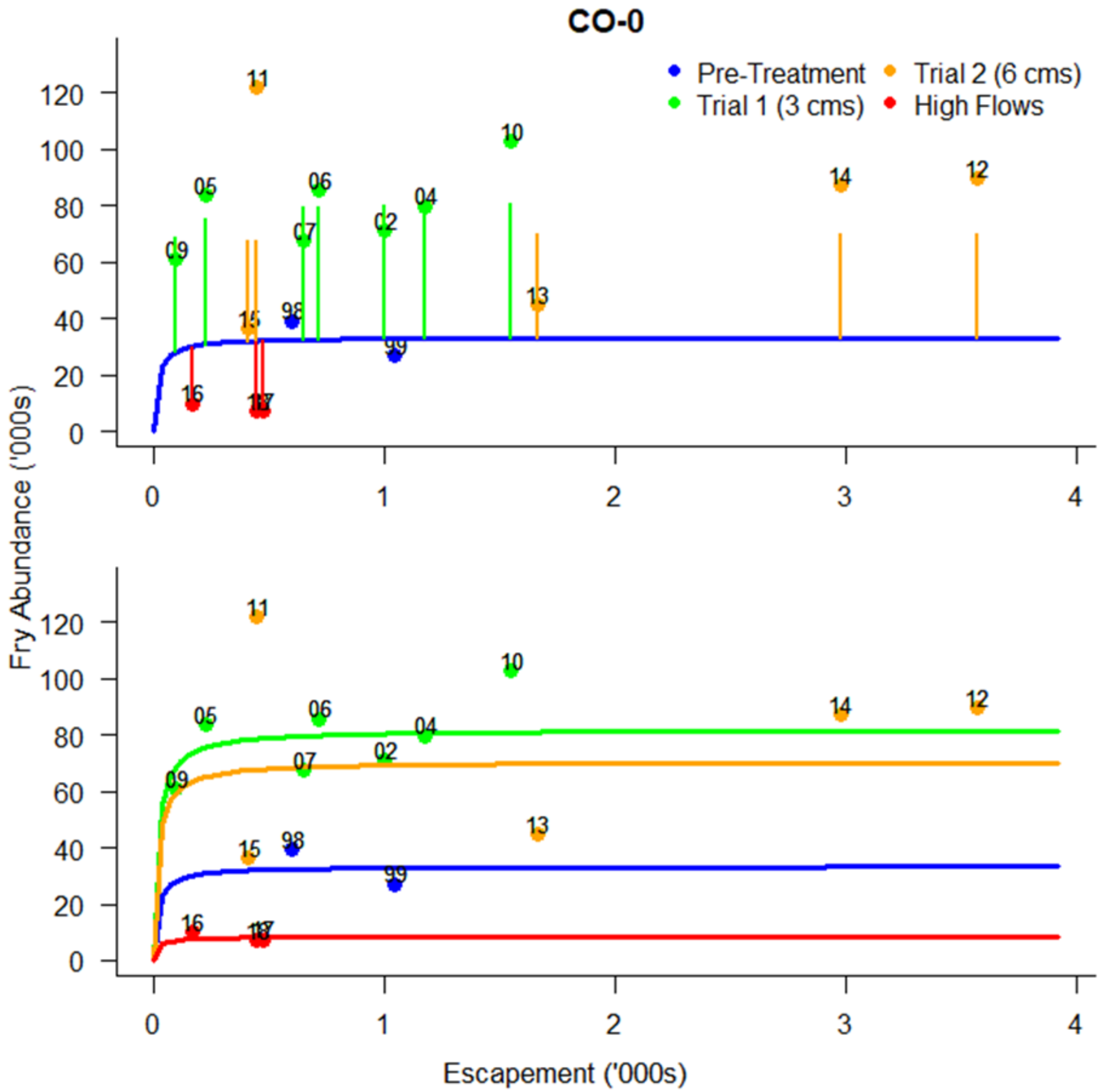


Figure 3.18 Spawner-fry coho Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 375 fry/spawner (50% egg-fry survival rate). Points show annual estimates of escapement and Age-0+ abundance with the label beside each point showing the recruitment year. The blue line in the top plot shows the base stock-recruitment curve under pre-flow conditions (Trial 0). The vertical lines in the top plot show the shift of the base stock-recruitment curve for the other three flow treatments. The bottom plot shows the treatment-specific stock-recruitment curves (e.g. the curve that results from drawing a line through the ends of the vertical lines in the top plot).

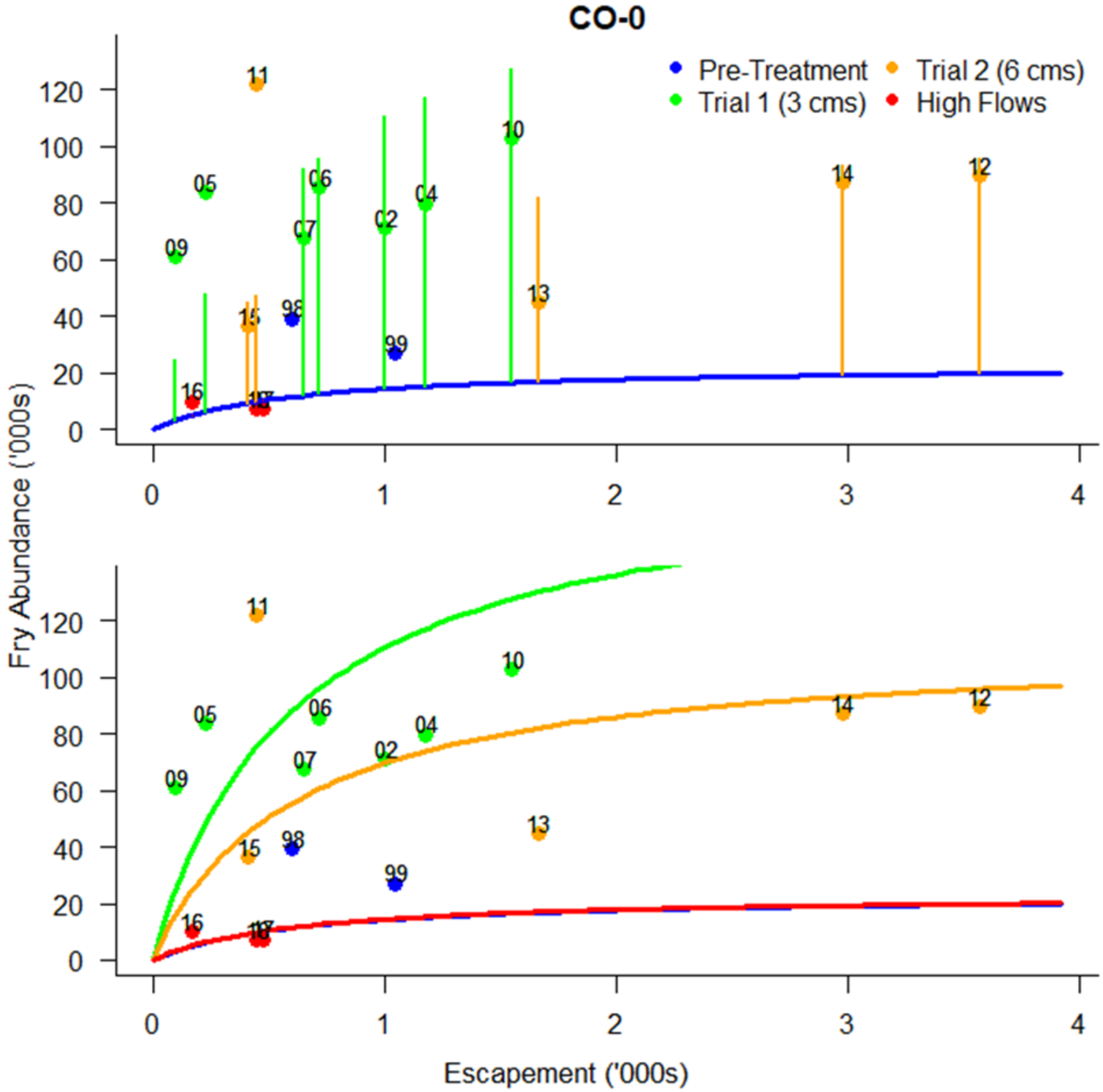


Figure 3.19 **Spawner-fry coho Beverton-Holt stock-recruitment curves fit assuming a maximum initial slope of 37.5 fry/spawner (5% egg-fry survival rate). See caption for Figure 3.18 for additional details.**

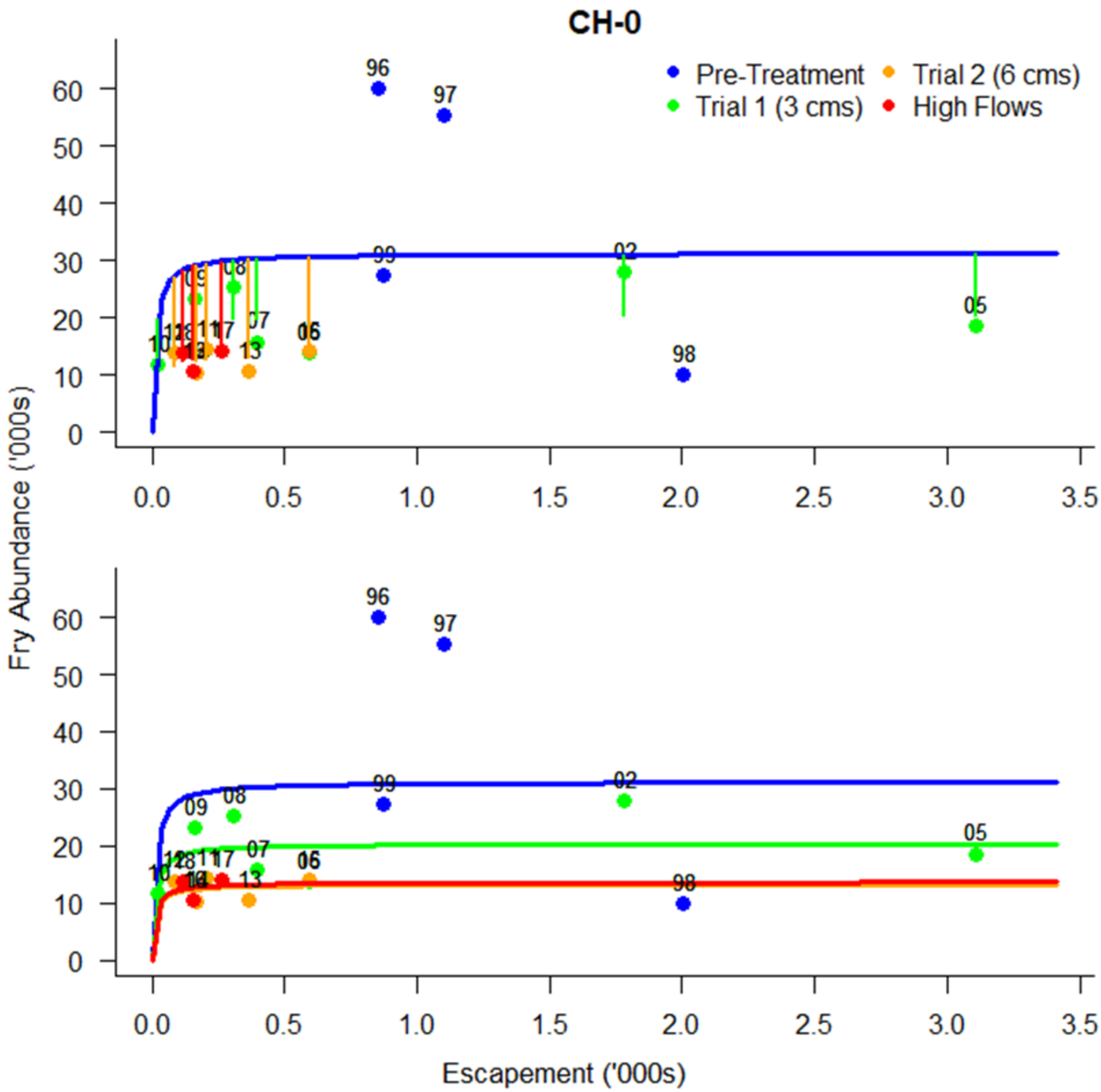


Figure 3.20 Spawner-fry chinook Beverton-Holt stock-recruitment curves fit with a constraint that assumes a maximum egg-fry survival rate of 50% (maximum initial slope of 1250 fry/spawner). See caption for Figure 3.18 for details.

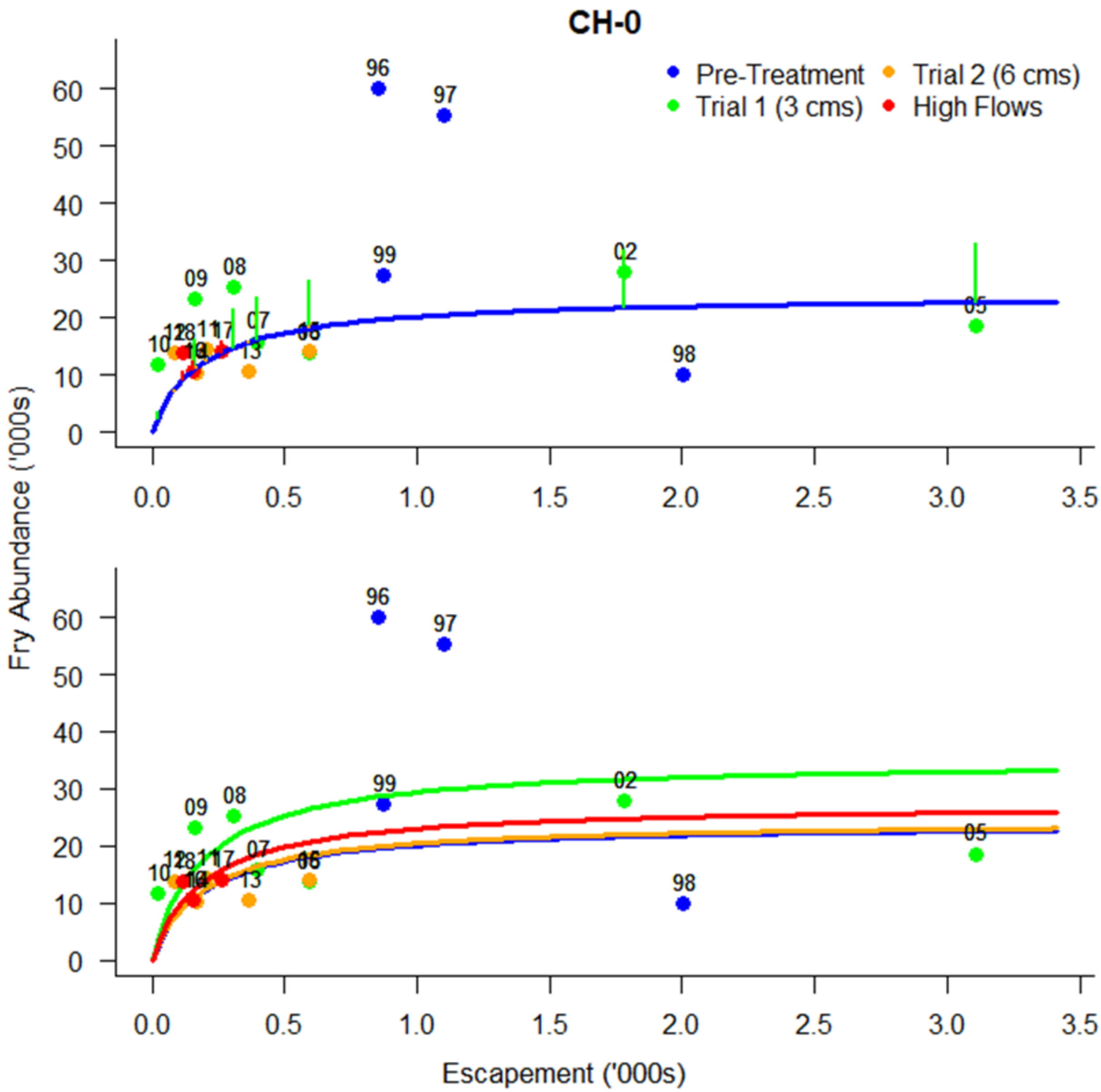


Figure 3.21 **Spawner-fry chinook Beverton-Holt stock-recruitment curves fit with a constraint that assumes a maximum egg-fry survival rate of 5% (maximum initial slope of 125 fry/spawner). See caption for Figure 3.18 for details.**

3.2. Modified Operations (High Flow) Monitoring

3.2.1. High Flow Monitoring

The modified operations (i.e., high flow release) period in 2018, along with a description of the discharge and continuous water temperature measurements at a range of stations downstream of the dam are provided in Section 3.1.1. Carpenter Reservoir was drawn down to approx. 615.3 m in 2018 (which was a similar low pool elevation to 2017 and 17.4 m lower than 2016; i.e., 615.2 m and 632.7 m, respectively). However, the reservoir was filling across the high flow period in 2018 from 618.1 m to 636.7 m, a total change of 18.6 m. The fill rate was highest between 6 and 22 May (average +0.90 m/day) and then diminished for the remainder of the survey period (23 May to 26 July) to an average of +0.07 m/day. Similar fill rates were observed in 2017, but occurred approx. 2 weeks later that year.

The following results were summarized from the 2018 Lower Bridge River High Flow Monitoring Summary Field Report prepared by Coldstream Ecology, Ltd. (O'Farrell and McHugh 2018) and associated data. For additional information, beyond what is included here, refer to that report.

Water Quality

Air and water temperatures generally increased in the LBR across the monitoring dates (there was good agreement between the spot measurements and the logger data), turbidity levels generally declined (i.e., highest levels were on the first date of the high flows – 11 May 2018), and total dissolved gas (TDG) levels generally increased as flows were initially ramped up between 11 and 24 May, and then stabilized across the remainder of the high flow period (Table 3.11).

Water temperatures were within the range of 9.9 to 16.5°C and were relatively consistent among locations on each survey date. The upper range of these temperatures were 3-4°C warmer than daily average temperatures for this period during trials 1 and 2 (as described in Section 3.1.1). These temperatures were within the optimal range for rearing by chinook, coho and steelhead, as cited in the literature, which may have contributed to the larger mean size of the juveniles from these species by the time of the annual stock assessment sampling in September (McCullough et al. 2001; Oliver & Fidler 2001; Myrick & Cech 2000).

Turbidity values increased from base levels (between 3.3 to 5.9 NTU, according to location) recorded on 1 March at a flow release of 1.9 $m^3 \cdot s^{-1}$, up to the highest recorded levels (between 35.9 and 38.0 NTU) on the first day of the high flow period as flows were ramped up from 15 to 26 $m^3 \cdot s^{-1}$, and Carpenter Reservoir filled from 618 m to 624 m elevation. Turbidity values then diminished across survey dates at each monitoring location before stabilizing between approx. 4 and 8 NTU by mid-June until the last survey date. The turbidity values in 2018 were lower than those recorded in 2017 (i.e., the 2017 peak was between 47.9 and 57.9 NTU on 9 June 2017, and lowest values were between 13.9 and 15.7 NTU on 29 June 2017, according to location). Turbidity values in 2018 were elevated above levels recorded at the Trial 2 peak flows (i.e., 15 $m^3 \cdot s^{-1}$) for a total of 15 days (i.e., 9 to 24 May).

The lowest TDG levels were 100% to 101% saturation at each site on the first date it was measured (9 May). As flow release levels increased, the TDG level at each site increased and then plateaued for the remainder of the survey period. At the top site (Terzaghi Dam Plunge Pool; Rkm 40.9) the TDG saturation increased to between 110% and 112%. At the middle site (Russel Springs; Rkm 30.4) TDG saturation increased to between 105% and 107%. At the lowest site (Upstream of the Yalakom; Rkm 25.5) TDG saturation increased to between 106% and 109%. To-date, the mechanism causing the increased TDG at high flows from the low-level outlet at Terzaghi Dam has not been determined, but the observed saturation levels have been below the thresholds for triggering a mitigation response according to Table 1 of BC Hydro's Total Dissolved Gas Management Strategy (i.e., BC Hydro 2014). Despite this, a low level assessment (under "kokanee entrainment monitoring") was conducted on each survey date to look for fish exhibiting signs of ill-effects, or mortalities.

Table 3.11 Summary of water quality measurements taken at 3 monitoring locations in the Lower Bridge River and 1 location in the Yalakom River across the high flow release period in 2018 (from O'Farrell and McHugh 2018).

Site / Parameter	Survey Date																				
	Mar	**May**							**Jun**								**Jul**				
	1	**6**	**9**	**11**	**18**	**24**	**29**	**31**	**5**	**8**	**12**	**14**	**18**	**22**	**26**	**28**	**3**	**5**	**13**	**19**	**26**
Carp. Res. Elevation (m)	629	618	622	624	630	633	634	635	635	635	635	634	634	635	636	636	636	636	636	636	637
TRZ Release ($m^3 \cdot s^{-1}$)	2	14	15	26	45	70	72	72	72	72	72	86	100	101	102	102	102	83	67	44	27
TRZ Plunge Pool (Rkm 40.9)																					
Water (°C)	-	9.9	11.0	9.5	10.5	11.4	11.3	11.2	11.1	11.7	11.9	12.2	12.0	11.9	12.6	12.6	13.1	13.3	14.1	13.8	14.4
Air (°C)	-	17.8	19.0	22.7	21.3	25.6	16.6	15.4	16.5	18.4	18.3	18.3	27.4	17.8	19.1	17.8	23.1	23.0	22.7	24.4	27.5
TDG (%)	-	-	101	108	110	112	111	110	111	111	112	111	112	111	111	112	111	109	110	111	110
Turbidity (NTU)	5.3	29.3	27.1	37.6	32.3	24.3	18.8	17.5	13.8	11.5	7.3	6.3	5.7	4.7	4.4	4.9	6.5	6.2	7.1	7.7	8.1
Conductivity (µS/cm)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	76.0	76.0	75.0
Alkalinity (mg/L)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	35.0	30.0	35.0
Russel Springs (Rkm 30.4)																					
Water (°C)	-	11.0	10.9	9.5	10.9	11.8	11.6	11.6	11.6	12.2	12.1	12.6	12.7	12.5	13.1	13.1	13.8	14.2	14.8	14.9	15.6
Air (°C)	-	21.5	18.4	19.5	21.6	23.3	14.1	13.7	15.5	18.8	17.6	17.6	30.1	20.5	18.1	18.6	25.9	23.9	25.4	26.3	31.1
TDG (%)	-	-	101	105	104	106	106	105	107	106	106	105	105	106	106	106	105	106	106	106	105
Turbidity (NTU)	5.9	23.1	23.5	38.0	30.4	23.2	19.0	17.3	14.3	11.4	8.7	7.8	6.2	5.0	4.6	4.9	6.5	6.5	6.9	8.1	7.3
Conductivity (µS/cm)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	78.0	82.0	80.0
Alkalinity (mg/L)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	35.0	35.0	35.0
U/S Yalakom (Rkm 25.5)																					
Water (°C)	-	11.9	11.5	9.9	11.3	12.1	11.7	12.0	11.7	12.5	12.3	13.0	13.3	12.8	13.7	13.5	14.0	14.6	15.3	15.5	16.5
Air (°C)	-	23.8	23.5	20.2	24.4	25.2	16.1	15.5	18.0	21.8	19.2	21.8	34.4	22.5	23.4	22.9	23.8	27.1	27.1	30.0	35.1
TDG (%)	-	-	100	104	104	106	106	106	107	109	106	106	106	107	107	106	106	107	107	107	106
Turbidity (NTU)	3.3	28.5	23.0	35.9	31.2	24.4	18.5	17.6	14.7	13.9	7.4	7.8	8.4	6.6	6.8	5.5	6.8	6.8	6.9	7.5	7.1
Conductivity (µS/cm)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	79.0	83.0	83.0
Alkalinity (mg/L)	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	40.0	40.0	40.0
Yalakom River																					
Turbidity (NTU)	-	-	-	-	-	-	5.7	4.0	2.0	1.9	1.6	1.7	3.7	18.0	5.9	3.3	2.9	2.7	2.7	2.6	1.8

Conductivity and alkalinity were measured at each LBR mainstem site on the last three survey dates. At the high flows in 2018, values were relatively consistent among the sites (range was 75 to 83 µS/cm for conductivity, and 30 to 40 mg/L CaCO3 for alkalinity). For reference against values for these parameters from past flow treatments, the mean conductivity and alkalinity by trial and season for similar locations in reaches 3 and 4 are provided in Table 3.12. The flow release from Carpenter Reservoir is characterized by lower conductivity and alkalinity than the groundwater and tributary inflows to the LBR channel. Therefore, following the onset of the continuous flow release, and particularly when the flow release magnitude has dominated flow volumes (i.e., the Trial 3 high flows), the values for these parameters have tended to decrease and become more consistent among stations, particularly above the Yalakom confluence. When measured again as a part of routine monitoring at lower flows in the fall of 2018, alkalinity values ranged from 29 to 50 mg/L CaCO3 from the top of Reach 4 to the bottom of Reach 3. Conductivity was not measured in the fall.

Table 3.12 Summary of mean conductivity and alkalinity values for Site A in Reach 4 and sites D and E in Reach 3 by flow trial and season.

Parameter	Index Site (Rkm)	Trial 0			Trial 1			Trial 2	Trial 3
		Spring	Summer	Fall	Spring	Summer	Fall	Fall	Fall
Conductivity (µS/cm)	A (39.9)	-	-	-	95	85	71	79	-
	D (30.4)	179	200	220	122	119	98	111	-
	E (26.4)	178	194	216	114	113	101	114	-
Alkalinity, as $CaCO_3$ (mg/L)	A (39.9)	-	-	-	76	67	56	41	28
	D (30.4)	143	165	184	92	94	77	60	43
	E (26.4)	147	170	182	93	92	81	62	49

Kokanee Entrainment

Unlike in the past two high flow years (2016 and 2017), entrained kokanee were not observed within the 1.5 km survey area below the dam in 2018. Twenty surveys were conducted from 6 May to 26 Jul on each of the dates summarized in Table 3.11 (for water quality sampling, above), which bracketed the high flow release period. As noted in past high flow monitoring reports, effective enumeration of fish in the river from shore was hampered by the high flows and turbidity levels, which made for poor visibility conditions into the water.

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

Bank Erosion and Sediment Recruitment

Nineteen bank erosion sites were assessed during the high flow period in 2018. Fourteen of these sites had been pre-selected based on past assessments in reaches 3 and 4, and five additional sites were identified by field crews during their weekly monitoring activities. Seventeen sites were on river right and two were on river left. All sites, except one, appeared to be comprised of naturally-derived substrate materials; the river left site at Fraser Lake (Rkm 33.3) was comprised of road-based materials, and was located just upstream of the rip rap placed at this site to address erosion issues in 2016. See the map produced by Coldstream Ecology Ltd. in Appendix C for the erosion monitoring site locations.

Twelve of the nineteen sites showed signs of erosion and substrate recruitment to the river from the 2018 high flows (Table 3.13). In most cases the erosion of material was caused by the interaction of the widened river channel (at high flows) with the base of an active alluvial slide area adjacent to the river (Photo 3.1). Two erosion sites were classified as covering a large area (>1000 m^2), and both occurred in Reach 2. The largest area, named “Below Horseshoe” at Rkm 22.5, was approximately 3000 m^2 and the other was estimated at 1760 m^2 in the Horseshoe Bend (Photo 3.1). Two medium-sized erosion sites were estimated to cover approximately 255 m^2 and 275 m^2 (sites #234 and #250); and eight sites were classified as small, covering a total of 63 m^2 altogether. The remaining seven assessed sites (i.e., Plunge Pool, #238, Rkm 35.0, Fraser Lake, #242, #244b and #249) did not show any signs of bank erosion or deposition under the 2018 high flows.

As noted in the Coldstream report, many of the eroded areas became more visible as water levels diminished from peak levels, suggesting that the majority of substrate recruitment occurred at or below the high flow waterline at these locations. Due to poor visibility related to the high flows and turbidity during the monitoring period, assessment of substrate deposition within the river channel could not be completed. Following the first rampdown from peak flows on 4 July 2018 (i.e., 102.0 to 82.6 $m^3 \cdot s^{-1}$), no further changes to erosion sites were evident.

Table 3.13 Summary of erosion and deposition sites observed during surveys at high flows in 2018 (O'Farrell and McHugh 2018).

Location Names	UTM Coordinates (Zone 10U)	River Bank (L or R)	Reach	Approx. Length (m)	Approx. Width (m)	Approx. Area (m^2)	Initial Observed Discharge (m^3/s)	Sediment Composition (%)			
								Boulder	Cobble	Gravel	Fines
234	555498E 5626306N	R	4	51	5	255	15		20%	80%	
236	555928E 5626290N	R	4	1	1	1	100	10%	80%		10%
239	558144E 5627498N	R	3	2	1	2	100		10%	90%	
243	558136E 5630037N	R	3	8	2	16	70	5%	15%	80%	
244	558070E 5630204N	R	3	2	1	2	85		20%	80%	
245	557879E 5630327N	R	3	30	1	30	70	10%	10%	80%	
246	557921E 5631579N	R	3	4	1	4	45	5%	20%	75%	
247	556492E 5631464N	R	3	5	1	5	70	5%	20%	75%	
248	556450E 5631645N	R	3	3	1	3	100	10%	40%	50%	
250	555909E 5632828N	R	3	55	5	275	26.4	5%	5%	90%	
Horseshoe Bend (Top Corner)	559302E 5634649N	R	2	220	8	1760	26.4	5%	15%		80%
Below Horseshoe	559608E 5634038N	R	2	300	10	3000	26.4		20%		80%

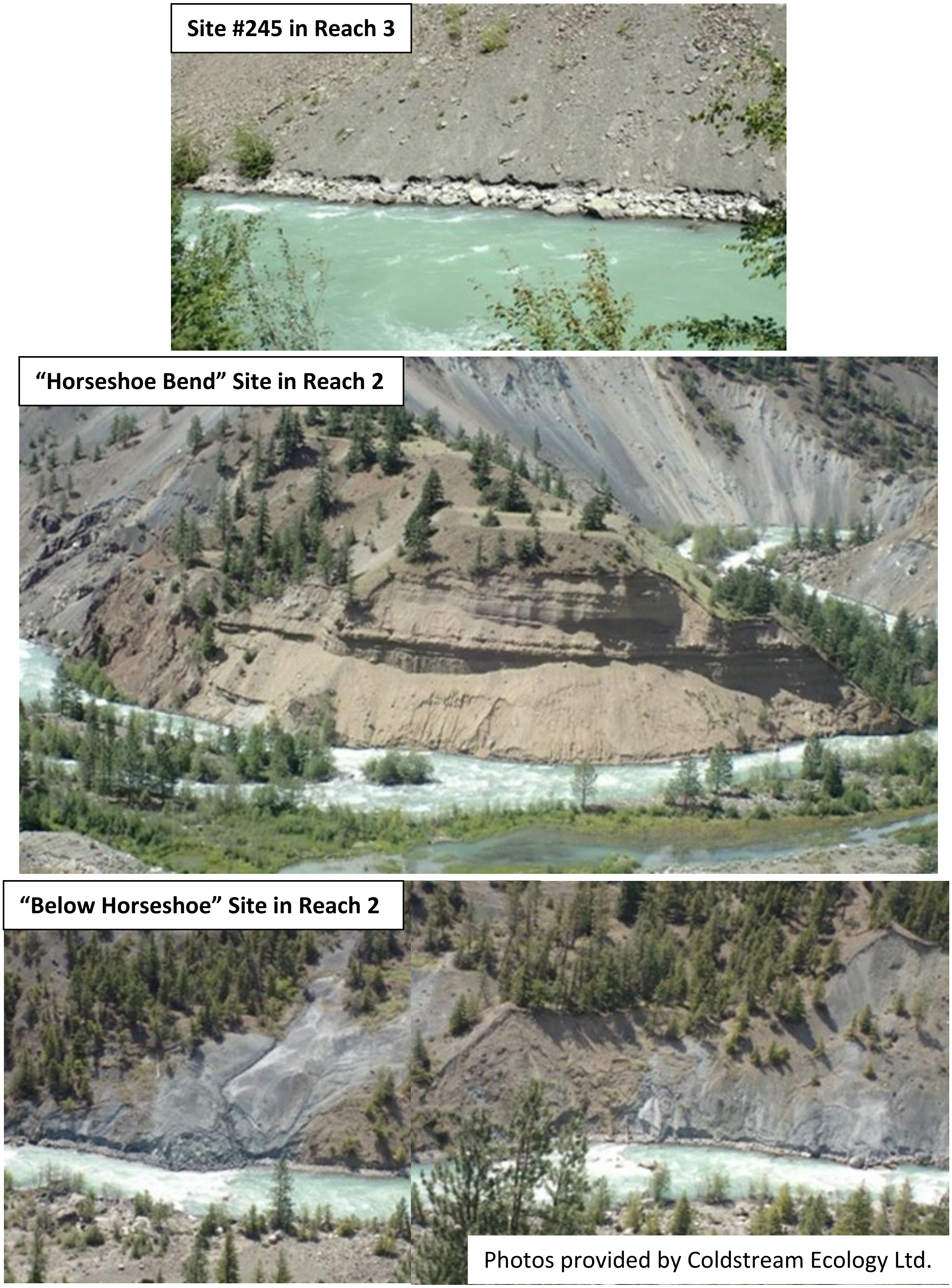


Photo 3.1 Examples of bank erosion at 3 sites in the Lower Bridge River at the interface between the widened river channel and the base of active alluvial slides.

Fish Stranding Site Reconnaissance

A summary of the high flow fish stranding site reconnaissance survey results is provided in Table 3.14, and a map of the locations produced by Coldstream Ecology Ltd. is provided in Appendix D. Eighteen potential stranding locations were identified between the Terzaghi Dam plunge pool and the Applesprings off-channel habitat in Reach 1. Six sites were in Reach 4, seven sites were in Reach 3, three sites were in Reach 2, and two sites were in Reach 1. Eight of the sites were on river left side, and the other ten were on river right. Potential stranding risk was qualitatively rated as Medium to High for all of the sites (according to the criteria provided in Section 2.2.1). The range of flows from the release at Terzaghi Dam when initial dewatering was expected to occur at the identified locations spanned from 82 to 27 $m^3 \cdot s^{-1}$; a higher proportion were at flows below 55 $m^3 \cdot s^{-1}$ (65%) than above (35%). Photos of the reconnaissance sites are not included in this report, but are available on request from Coldstream Ecology, Ltd.

Table 3.14 Summary of potential fish stranding locations identified during high flow reconnaissance monitoring in 2018 (O'Farrell and McHugh 2018). Note: "Flows when identified" were assessed on the ascending limb of the hydrograph, and "Flows at Initial Dewatering" were assessed on the descending limb.

Location Names	River Bank (L or R)	Reach	River Km	Flows when Identified ($m^3 \cdot s^{-1}$)	Flows at Initial Dewatering ($m^3 \cdot s^{-1}$)
Plunge Pool	L	4	41.0	45	45
Rkm 41	R	4	41.0	45	55
Below Plunge Pool	R	4	40.1	25	67
Rkm 38.4	R	4	38.4	45	55
38.3	R	4	38.3	25	27
37.9	R	4	37.9	85	55
35.0	R	3	35.0	25	27
31.4	L	3	31.4	85	82
30.6	L	3	30.6	75	55
Russel	L	3	30.4	45	67
Rkm 30.3	R	3	30.3	45	55
29.9	L	3	29.9	25	55
Below Hell Bar	R	3	28.6	45	67
Below Horseshoe (1)	L	2	22.0	25	82
Below Horseshoe (2)	L	2	22.0	25	55
Above Camoo	L	2	20.2	44	55
Little Horseshoe	R	1	19.0	55	45
Above Applesprings	R	1	16.4	25	67

3.2.2. Juvenile Salmonid Habitat Availability and Displacement

A set of pre-selected sites were sampled for juvenile salmonids across a range of flow release levels on the ascending limb, peak, and descending limb of the 2018 hydrograph. Sites were open and sampled in a single pass so there was no way of determining differences in capture probability at the various flow levels, but we divided the 1-pass catch by the area sampled to generate relative densities for each species and age class by site and flow to facilitate some comparisons (Figure 3.22 and Figure 3.23).

Coho fry densities (Figure 3.22, left-side plots) were highest in off-channel habitats (high quality), followed by mainstem habitats pre-designated as low quality, and then mainstem habitats pre-designated as high quality, at the lowest flows on the ascending limb of the hydrograph (i.e., 15 and 28 $m^3 \cdot s^{-1}$). Once flows were ramped up past 28 $m^3 \cdot s^{-1}$, coho fry densities dropped right off at all high quality and low quality sites for the remainder of the surveys (other than a small bump at the high quality mainstem site near Applesprings (Rkm 11.3) and the "Applesprings Lower" off-channel site (AS_Low) in Reach 1 at 67 $m^3 \cdot s^{-1}$ on the descending limb of the hydrograph).

Chinook fry densities (Figure 3.22, right-side plots) were lower than coho to start with, but similarly tended to be highest at various high and low quality sites (mainstem and off-channel) at the lowest discharges on the ascending limb of the hydrograph (i.e., 15 and 28 $m^3 \cdot s^{-1}$). Also similar to coho, the mainstem densities dropped off above 28 $m^3 \cdot s^{-1}$, and stayed low for the remainder of the surveys in each sampled habitat type.

Mykiss fry tended to show a different pattern (Figure 3.23, left-side plots). Their densities were low in all habitat types across the ascending limb and peak of the hydrograph, and then increased on the descending limb, particularly in the mainstem sites. This pattern likely has to do with the emergence timing for the new year-class of this species. Steelhead and rainbow trout tend to spawn in April and May in the LBR, so the fry likely emerge during the high flow period. Based on these figures, it appears that temperature conditions under the high flows were suitable for mykiss incubation, and the fry were starting to emerge sometime around the early to mid July surveys at 82 and 67 $m^3 \cdot s^{-1}$, which was similar to the timing noted under the previous flow trials. Highest abundances were on the 67 and 27 $m^3 \cdot s^{-1}$ surveys on the descending limb of the hydrograph at high and low quality mainstem sites. Densities in the off-channel habitats tended to be low across the survey period, with only a slight increase on the last date (27 July 2018) at 27 $m^3 \cdot s^{-1}$.

Mykiss parr densities (Figure 3.23, right-side plots) tended to be highest in the off-channel sites, and on the ascending limb of the hydrograph for all habitat types. They appeared more tolerant of high flows at some of the sampled sites (i.e., the Bluenose outflow channel (BN Outflow), Rkm 30.4 HQ, and Rkm 29.0 LQ), where densities peaked at 70 $m^3 \cdot s^{-1}$. Above this discharge rate, parr densities dropped right off and then remained low across the remaining survey dates.

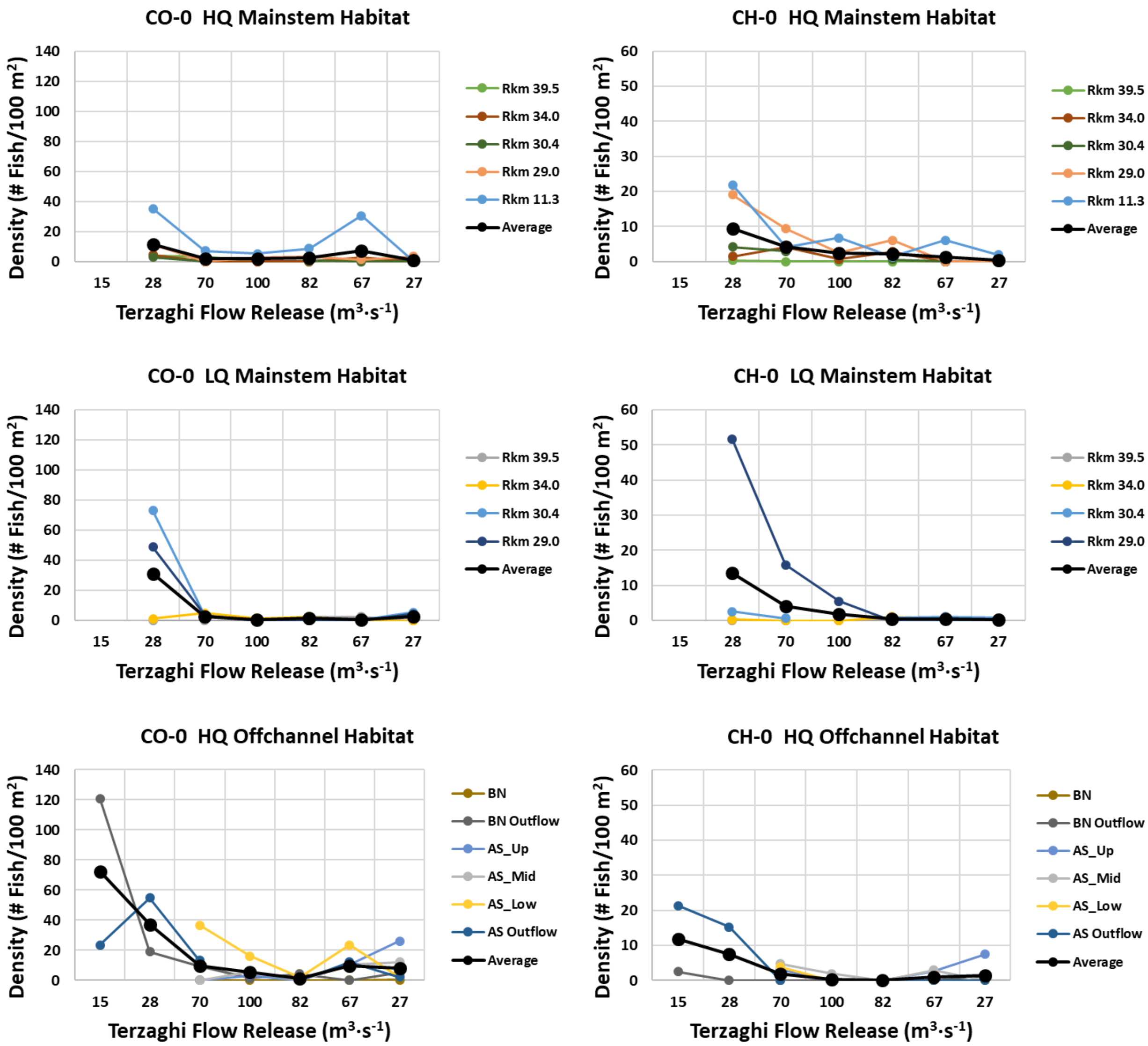


Figure 3.22 Relative densities of coho fry (left panels) and chinook fry (right panels) across a range of high flows in 2018 by habitat type: high quality mainstem (top row), low quality mainstem (middle row), and high quality off-channel (bottom row; BN = Bluenose; AS = Applesprings).

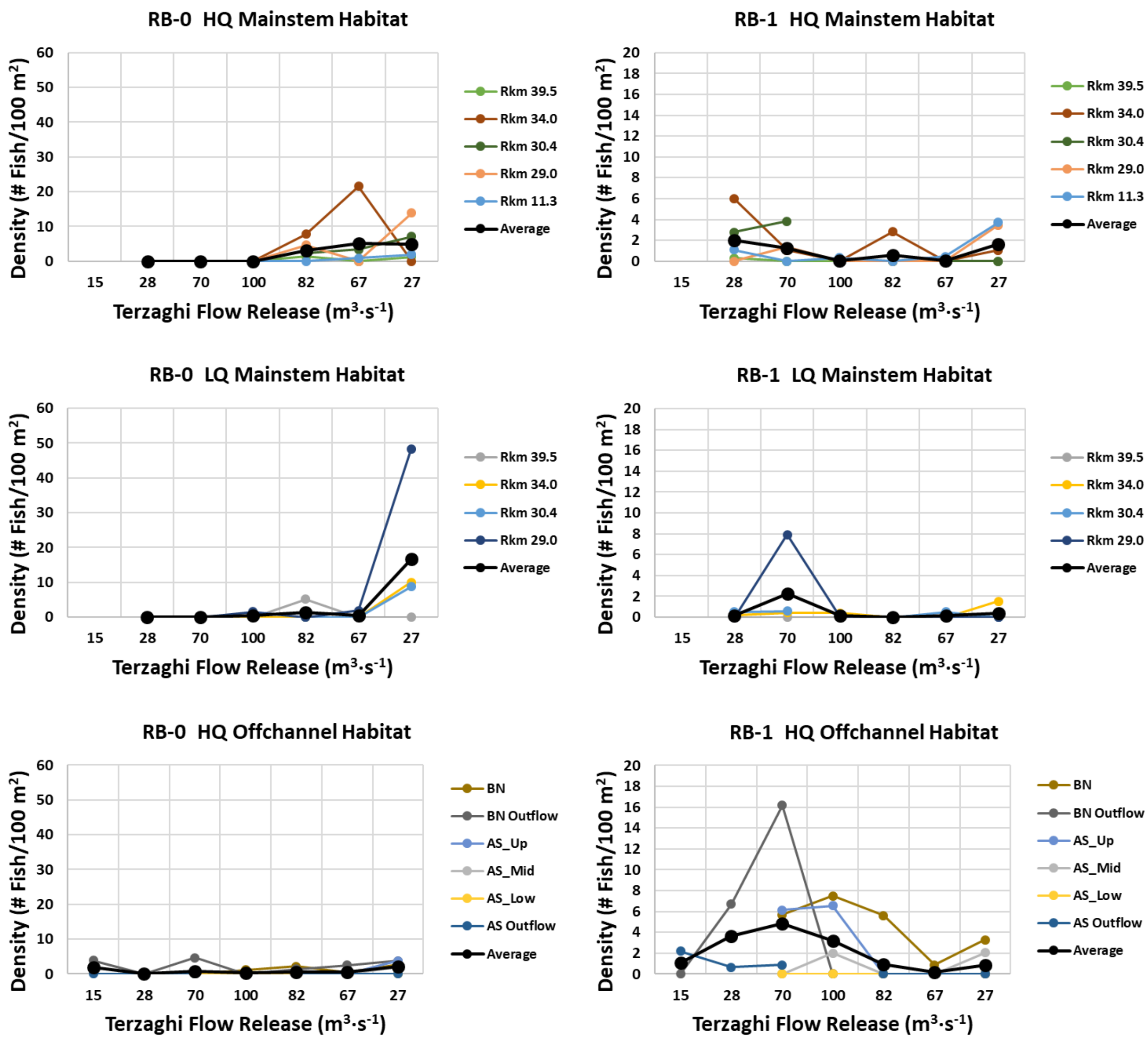


Figure 3.23 Relative densities of juvenile mykiss (Age-0+ left panels; Age-1 right panels) across a range of high flows in 2018 by habitat type: high quality mainstem (top row), low quality mainstem (middle row), and high quality off-channel (bottom row; BN = Bluenose; AS = Applesprings).

A summary of mean depths and velocities measured at each sampling site across the range of survey dates is shown in Figure 3.24. There was very little pattern apparent in the depth and velocity data among sites, other than that the high quality mainstem sites seemed a bit more responsive (or susceptible) to the effects of the high flows than the low quality mainstem sites. Depths and velocities at high quality mainstem sites ranged between 0.1 – 0.7 m and 0.02 – 0.78 m/s, respectively; whereas they ranged between 0.1 – 0.5 m and 0.00 – 0.40 m/s, respectively, at low quality mainstem sites. Depths and velocities were the most consistent at sites within the off-channel habitats (except for depths at the Applesprings outflow) which are not as directly influenced by the flow changes in the mainstem.

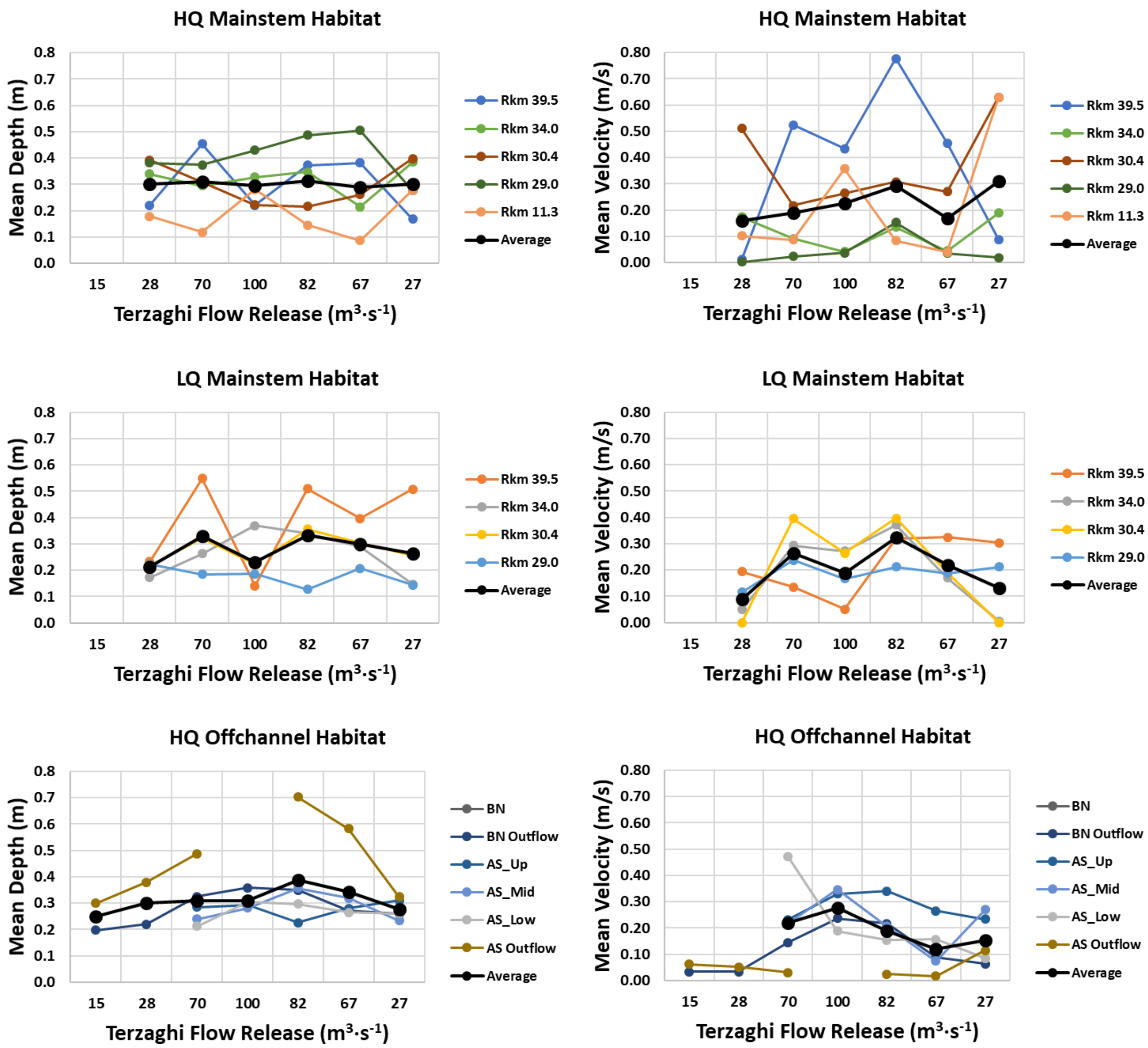


Figure 3.24 Mean depths (left panels) and mean velocities (right panels) at sites sampled across a range of high flows in 2018 by habitat type: high quality mainstem (top row), low quality mainstem (middle row), and high quality off-channel (bottom row; BN = Bluenose; AS = Applesprings).

Wetted area of the sampled sites for each habitat type across the range of flows are provided in Figure 3.25. These data reveal that while the sample-able area varied among sites at the different flows, the averages among sites for each type were relatively consistent, reflecting that while habitat area may be reduced at one location under a particular flow release magnitude, the loss may be compensated by increased area at another location and vice versa. In fact, the total site area (all types combined) varied by only 356 m^2 among all flows surveyed, between a maximum of 2,782 m^2 documented at the 67 $m^3 \cdot s^{-1}$ release to a minimum of 2,426 m^2 at the 27 $m^3 \cdot s^{-1}$ release. The same result was also apparent for the measured depths and velocities at the sampled sites (Figure 3.24), similarly suggesting that while suitable habitat conditions (defined according to HSI predictions for each species and age class – curves available from BC Hydro upon request) became less available at one location, the loss may have been compensated at least to some extent by improved conditions at another. However, this conclusion is tenuous due to the limited nature of this pilot sampling effort. BC Hydro's Telemac2D model would likely be a more powerful tool for predicting changes in rearing habitat area across the high flow range at the reach and study area scale. This has been included as a recommendation in Section 5.

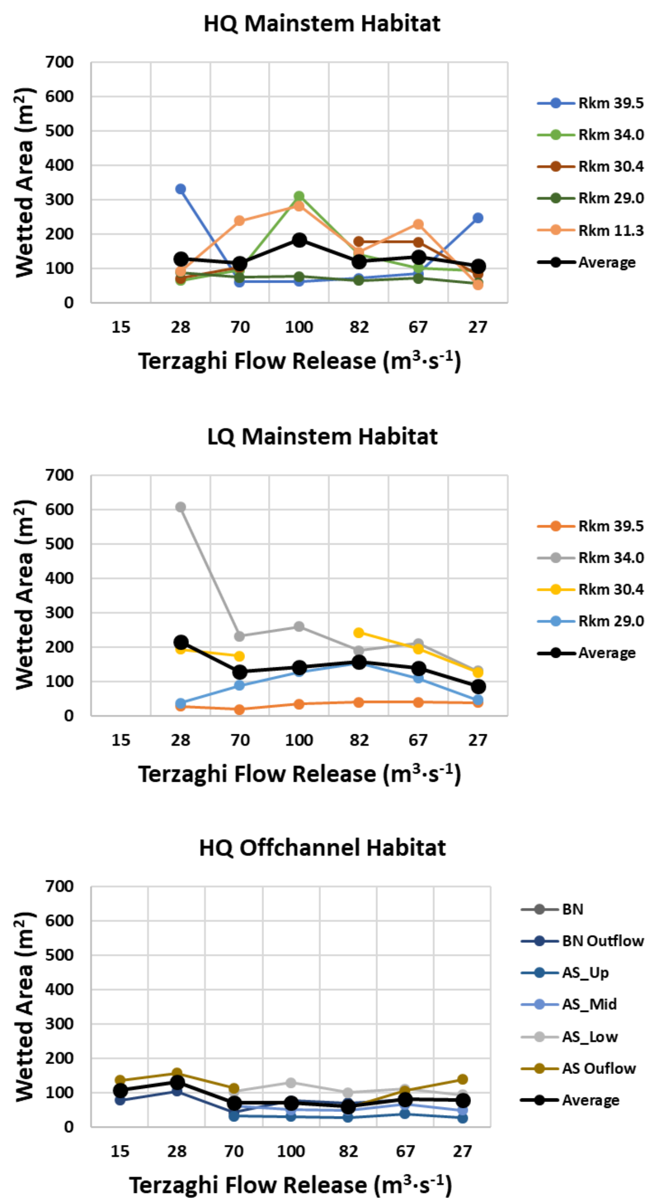


Figure 3.25 Wetted area of sampled extent at pre-selected sites across a range of high flows in 2018 by habitat type: high quality mainstem (top), low quality mainstem (middle), and high quality off-channel (bottom; BN = Bluenose; AS = Applesprings).

3.2.3. High Flow Ramp Down Monitoring and Stranding Risk Assessment

In the tables and figures throughout this section, comparable ramping information from the 2016 and 2017 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 2018 results, for reference.

Ramp downs from high flows (i.e., between 102 and 15 $m^3 \cdot s^{-1}$) occurred across 8 dates between 4 July and 1 August 2018, representing a total flow reduction of 86.9 $m^3 \cdot s^{-1}$ across that period (Figure 3.26 and Table 3.15). For more detailed information on flow and stage changes for each rampdown event, refer to Tables E1 to E3, and Table E5 in Appendix E. Total stage change at the 36.8 km compliance location was 122 cm, and maximum daily stage change rate implemented was 4.0 cm/hr. The implementation of some higher ramp rates in 2017 and 2018 (compared to past years) meant that the reduction of flows from a higher magnitude could be completed over a shorter timeframe (i.e., fewer hours of ramping per day). Flow ramping within the Trial 2 flow range (≤15 $m^3 \cdot s^{-1}$) was conducted over an additional 7 dates in August and 2 dates in October, 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).

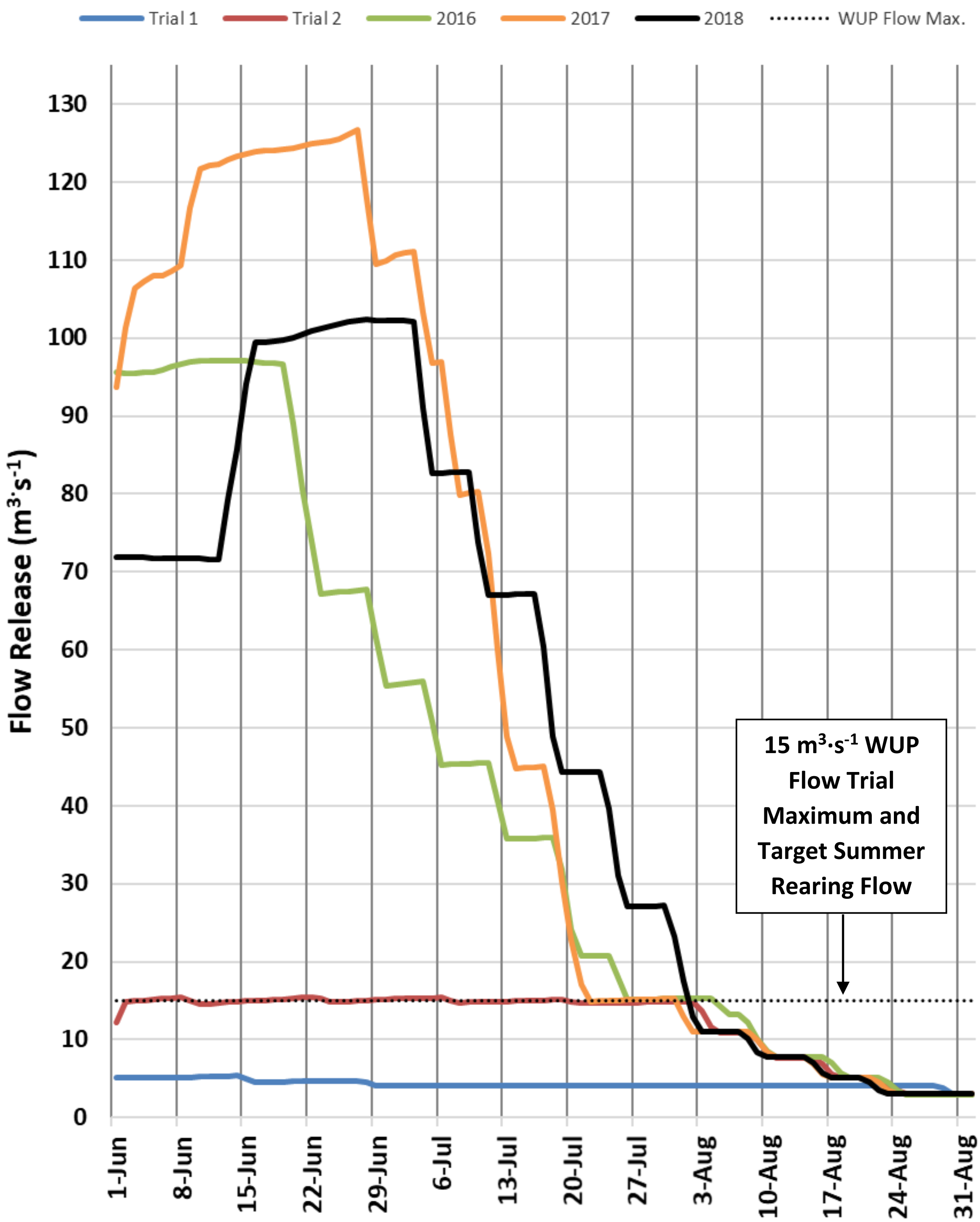


Figure 3.26 Schedule of flow releases and ramp downs from the peak period to the start of the fall low flow period in 2016, 2017 and 2018. For reference, Trial 1 and 2 flow releases are shown for the same period.

Table 3.15 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, 2017 and 2018. For more details on individual events refer to the tables provided in Appendix E.

Period	Year	Month(s)	# of Ramping Days	Total Flow Reduction ($m^3 \cdot s^{-1}$)	Total Stage Change (cm)	Maximum Daily Rate (cm/hr)
High Flow Ramp Events (>15 $m^3 \cdot s^{-1}$)	2018	Jul	8	-86.9	-122	-4.0
	2017	Jun – Jul	9	-96.5	-143	-4.1
	2016	Jun – Jul	8	-81.4	-108	-2.3
"Normal" Ramp Events (≤15 $m^3 \cdot s^{-1}$)	2018	Aug, Oct	9	-13.6	-62	-2.0
	2017	Aug, Sep	9	-13.7	-67	-2.6
	2016	Aug, Sep	10	-13.8	-67	-3.0

Coupling the BC Hydro flow release records with the continuous river stage level recorded at 36.8 km (known as the compliance location for tracking ramp rates) enabled characterization of the discharge-stage relationship at that location (Figure 3.27).

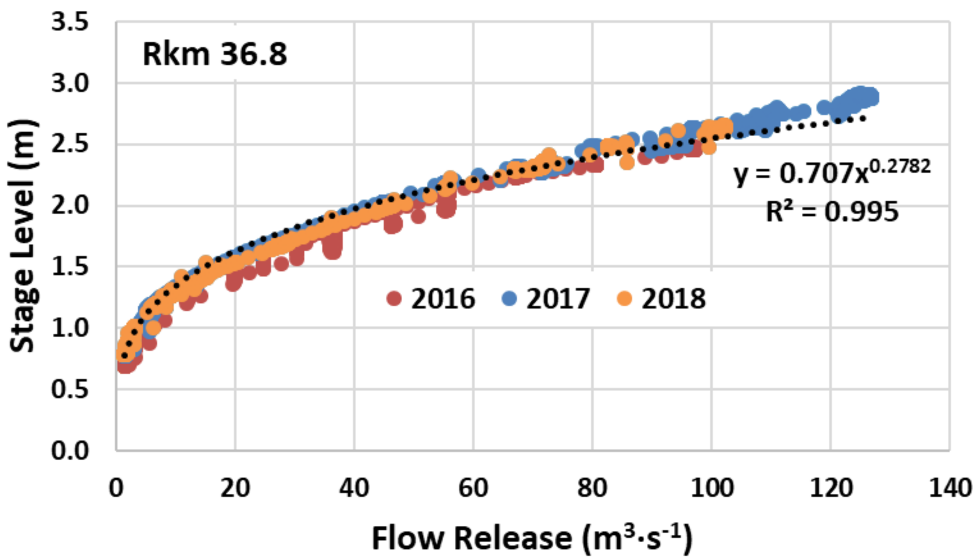
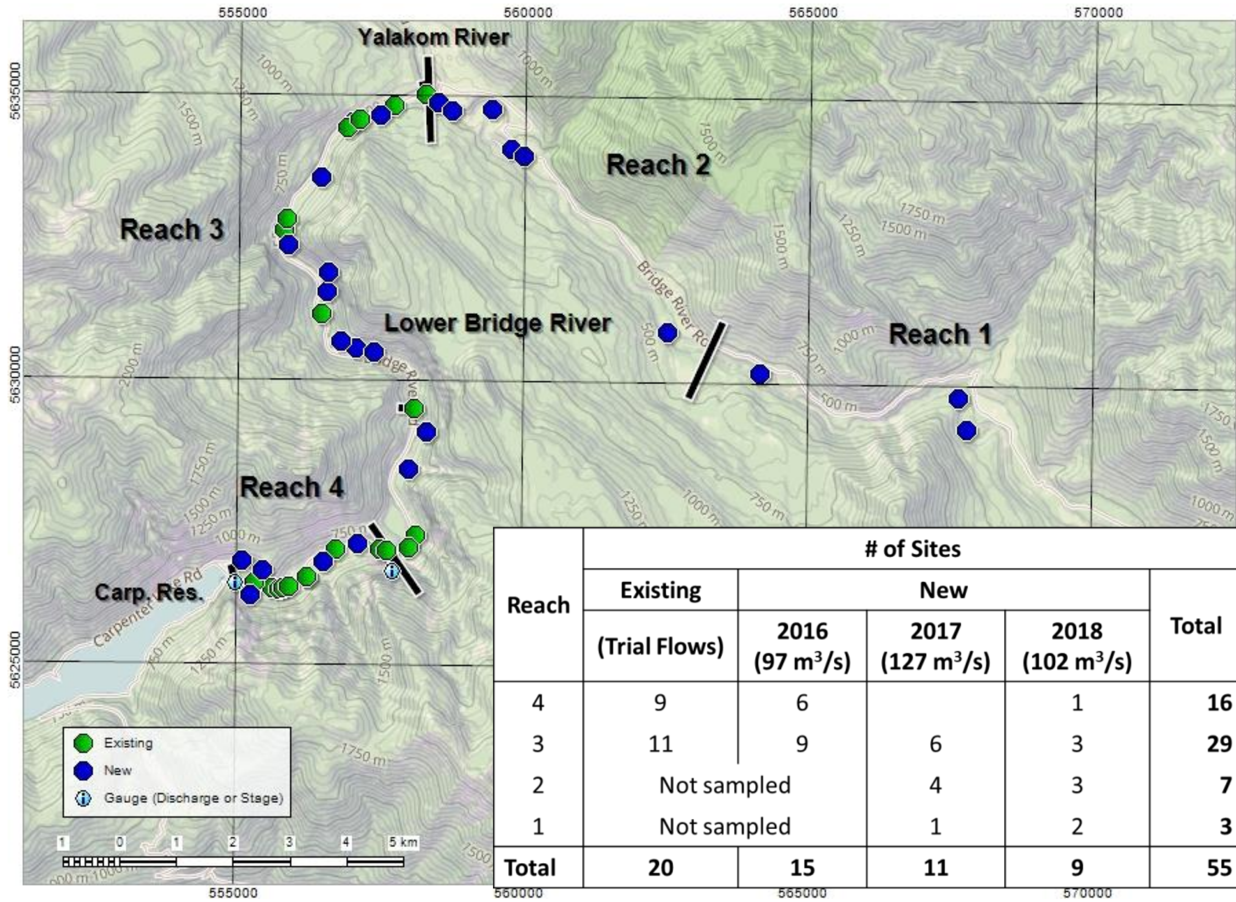


Figure 3.27 Discharge-stage relationship at 36.8 km (the compliance location) across the range of flows observed across all flow treatments. Separate data points for each high flow monitoring year (2016-2018) are shown.

The curve drawn through the points has a good fit (R^2 = 0.995), such that the associated equation ($y = 0.707x^{0.2782}$) may be useful for predicting stage changes for particular flow changes within this range. Stage values for discharges between 10 $m^3 \cdot s^{-1}$ and 60 $m^3 \cdot s^{-1}$ tended to be a bit lower in 2016, possibly due to some channel changes at the gauging location that have occurred with the high flows since then, so the current curve is based on the 2017 and 2018 data points. The curve may underestimate stage elevations for discharges >100 $m^3 \cdot s^{-1}$. It is clear from the relationship that the greatest degree of stage changes occurs at the lowest discharges (i.e., the initial slope is the steepest). Above ~10 $m^3 \cdot s^{-1}$ the slope begins to decrease, such that the discharge-stage relationship becomes close to linear across the higher flows.

As a result of the fish stranding site reconnaissance and flow ramp down surveys conducted during the three high flow years to-date, the incidence of fish stranding was documented at 35 new locations (*n*= 15 in 2016; *n*= 11 in 2017; and *n*= 9 in 2018) for flows >15 $m^3 \cdot s^{-1}$ across all four reaches of the Lower Bridge River (Figure 3.28). 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 (*n*= 25, or 71%) of the new sites at flows >15 $m^3 \cdot s^{-1}$ was in reaches 3 and 4. There have been 7 new sites added in Reach 2 and 3 new sites in Reach 1.



Reach	# of Sites Existing (Trial Flows)	New 2016 (97 m³/s)	New 2017 (127 m³/s)	New 2018 (102 m³/s)	Total
4	9	6		1	**16**
3	11	9	6	3	**29**
2	Not sampled		4	3	**7**
1	Not sampled		1	2	**3**
Total	**20**	**15**	**11**	**9**	**55**

Figure 3.28 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 from 2016 to 2018. Discharge and stage gauging locations are represented by the blue information symbol (i). Solid black lines represent the reach breaks. A table summarizing the number of sites is also included (inset).

Fish salvage numbers for the ramp downs across the high flow range were consistently low relative to the results for the Trial 2 range (≤15 $m^3 \cdot s^{-1}$; Figure 3.29). In previous years (≤2016), crews had noted incidental catches (fish salvaged before their strand-risky habitat had become isolated from the main channel flow or dewatered); however, for consistency with the 2017 and 2018 results, these incidental catches were not included in the analyses. However, it is important to note for 2018 that in cases where water temperatures were warming rapidly, causing stress to fish in habitats that were not yet fully isolated, crews elected to salvage fish as incidentals to reduce the incidence of mortality.

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 all high flow years to-date (see Section 3.1.5). Relative to the Trial 2 averages, abundance of coho and steelhead fry was down by 90% and 80%, 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.

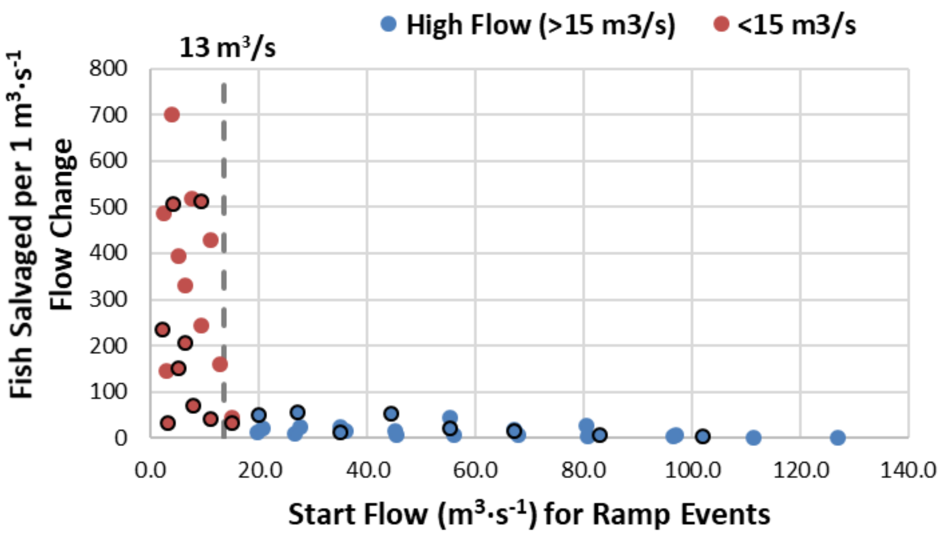


Figure 3.29 Relative differences in number of fish salvaged per increment of flow change for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) versus Trial 1 and 2 flows (≤15 $m^3 \cdot s^{-1}$). The vertical dashed line represents the approximate flow threshold (~13 $m^3 \cdot s^{-1}$) where the apparent break between high stranding risk and moderate or low stranding risk occurs. Note: values do not include incidental catches. Circles with black border represent 2018 data; Plain circles are data from past years.

Compared to survey results from the previous flow trials, relatively large areas of fish stranding habitat have been documented in 2016, 2017 and 2018 (66,892 m^2), primarily due to the addition of stranding site reconnaissance and salvage surveys in reaches 2 and 1 (Table 3.16). The proportions of stranding area by reach were 7%, 14%, 34%, and 45% for reaches 4, 3, 2, and

1, respectively. Under the trial flows (≤15 $m^3 \cdot s^{-1}$), the total stranding area was 16,548 m^2, which was more prevalent in Reach 3 than Reach 4 (70% and 30%, respectively).

Across the high flow range (>15 $m^3 \cdot s^{-1}$) in 2018, the highest proportion of salvaged fish per stranding habitat area was in Reach 3 (~8 fish per 100 m^2; Figure 3.30). The values for the other reaches were relatively small (≤3 fish per 100 m^2). Within the Trial 2 flow range (≤15 $m^3 \cdot s^{-1}$), fish stranding densities were greater and the highest proportion was in Reach 4 followed by Reach 3 (16 and 13 fish per 100 m^2, respectively). Note: These values are far lower than those documented for the ≤15 $m^3 \cdot s^{-1}$ flows under the previous flow treatments that were characterized by much higher juvenile salmonid abundance (i.e., Trials 1 & 2 means = 81 (range = 51 to 123) and 63 (range = 48 to 75) fish per 100 m^2 of salvaged area in reaches 3 and 4, respectively; Sneep 2016). Reaches 1 and 2 have not been surveyed within the trial flow range.

Table 3.16 Summary of fish stranding area and numbers of fish salvaged by reach for 2018 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	7	4,887 (7%)	125	3
	3	11	9,105 (14%)	710	8
	2	4	22,900 (34%)	551	2
	1	3	30,000 (45%)	413	1
High Flow Totals		**25**	**66,892**	**1,652**	**3**
Trial Flows (≤15 $m^3 \cdot s^{-1}$)	4	7	4,938 (30%)	792	16
	3	13	11,610 (70%)	1,470	13
	2	------ No data ------			
	1	------ No data ------			
Trial Flow Totals		**20**	**16,548**	**2,262**	**14**

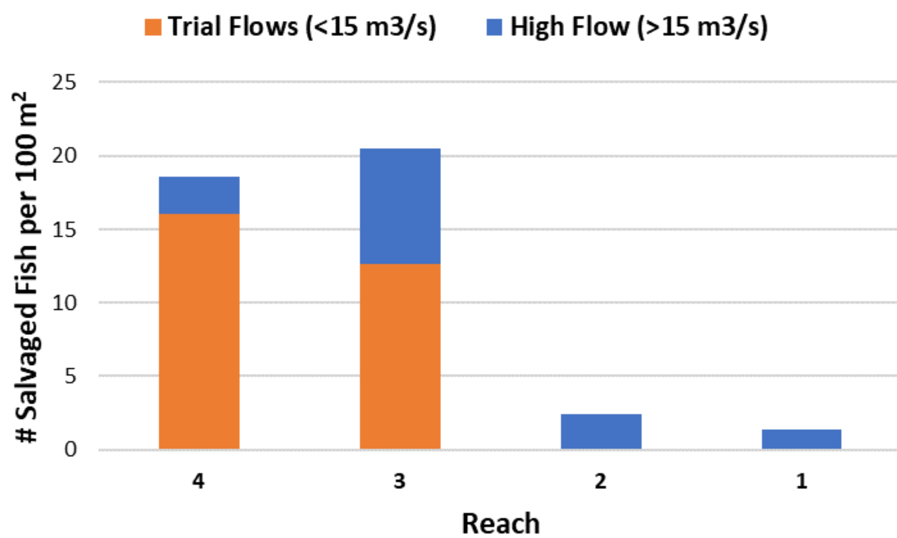


Figure 3.30 Summary of mean numbers of fish salvaged per dewatered habitat area by reach for 2018 high flow (>15 $m^3 \cdot s^{-1}$) and trial flow (≤15 $m^3 \cdot s^{-1}$) ranges. Note: there was no data for fish stranding or salvage in reaches 1 and 2 under the trial flows.

With the benefit of fish salvage crews on the ground, some higher ramp rates (up to 4.0 cm/hr) were implemented again in 2018 (as 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 sample size available from 2017 and 2018, the higher ramp rates employed for ramp downs within the high flow range (>15 $m^3 \cdot s^{-1}$) did not increase the incidence of stranding at the flow levels tested (Figure 3.31). 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, 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, 2017 and 2018, as mentioned above.

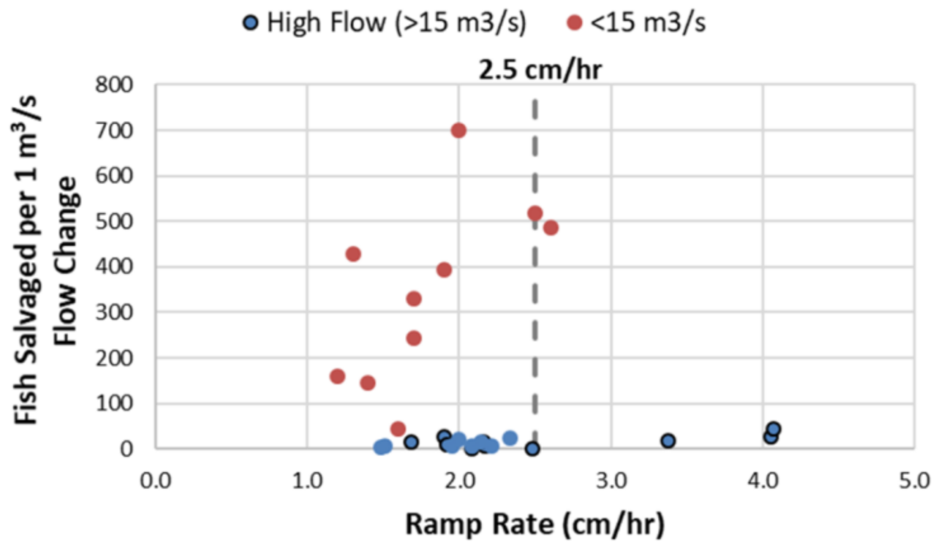


Figure 3.31 Relative incidence of fish stranding per increment of flow change according to different ramping rates under high flow (>15 $m^3 \cdot s^{-1}$) and trial flow (≤15 $m^3 \cdot s^{-1}$) ranges. The vertical dashed line depicts the ramp rate (2.5 cm/hr) specified in the WUP when fish salvage crews are not present. Circles with black border represent 2018 data; Plain circles are data from past years.

The proportions of identified stranding sites on river left (80%) versus river right (20%) were not equal under the trial flows (≤15 $m^3 \cdot s^{-1}$), even though both banks were accessible to fish salvage crews across a significant part of that range (Table 3.17). Note that these proportions are based on reaches 3 and 4 only as reaches 1 and 2 were not surveyed at flows below 15 $m^3 \cdot s^{-1}$. Across the high flow range (>15 $m^3 \cdot s^{-1}$), the distribution was closer to equal with 45% on river left and 55% on river right based on the new site reconnaissance conducted by staff from Coldstream Ecology Ltd. and Xwísten.

As identified for past fish salvage surveys under flow trials 1 and 2 (≤15 $m^3 \cdot s^{-1}$), coho and mykiss were the most frequently encountered species under high flows in 2018 (contributing 18% and 56% to the total catch, respectively; Table 3.18). The coho tended to be most prevalent at sites in reaches 2 and 1 under high flows, whereas mykiss were most abundant in reaches 4 and 3 under the trial flow range. This despite the fact that abundance of these fish was substantially reduced in 2018 overall, likely caused by the high flows. As noted in the Fish Stranding Protocol, coho and mykiss fry tend to be the most vulnerable to stranding because the habitat types preferred by this age class of these species (e.g., shallow edge areas and side channels/pools) are also among the habitat types that are most likely to dewater and result in fish stranding when flows are reduced. Fry may also remain in these habitats even as flows are dropping because they are less able to exploit deeper offshore areas where there are typically higher velocities, less cover, and increased risk of predation.

Table 3.17 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	
	n	%	n	%
Trial Flows (≤15 $m^3 \cdot s^{-1}$) *Reaches 3 & 4 only	16	80%	4	20%
High Flows (>15 $m^3 \cdot s^{-1}$) *New Site Reconn.	8	45%	10	55%
All	**24**	**63%**	**14**	**37%**

The least abundant of the target salmonid species in the salvage results were chinook fry, which were most abundant in reaches 4 & 3 within the trial flow range, and least abundant in those reaches under high flows. Chinook fry can occupy some of the same habitats as coho and steelhead fry, but they tend to be larger (because they emerge earlier in the year) so they can exploit habitats further from the river margins that are less likely to dewater. Also, they have been much less abundant in the study area overall since the flow trials began, and particularly in reaches 3 and 4 (see Section 3.1.5).

Table 3.18 Summary of numbers of fish salvaged by species-age class and reach under high flow ramp downs (>15 $m^3 \cdot s^{-1}$) in 2018.

Species	High Flows (>15 $m^3 \cdot s^{-1}$)		Trial Flows (≤15 $m^3 \cdot s^{-1}$)		Total
	Reaches 4 & 3	**Reaches 2 & 1**	**Reaches 4 & 3**	**Reaches 2 & 1**	
Chinook	24	93	130	-	247 (6%)
Coho	139	491	114	-	744 (18%)
Mykiss	539	373	1,349	-	2,261 (56%)
Other spp.	133	7	669	-	809 (20%)
All	**835**	**964**	**2,262**	**-**	**4,061**

Other species in the fish salvage catches were: bull trout (*n*= 10), O. nerka juveniles (*n*= 9), mountain whitefish (*n*= 7), sucker spp. (*n*= 13), redside shiner (*n*= 148), and sculpin spp. (*n*= 622). Of these, sculpin spp. were by far the most numerous, and most of these were salvaged on the ramp day #15 (22 Aug 2018) when flows were ramped from 4.1 to 3.0 $m^3 \cdot s^{-1}$. For shiners, the second most abundant "other spp.", the majority were salvaged on days #8 and #9 across the 20.0 to 11.0 $m^3 \cdot s^{-1}$ ramp range. For the specific catch totals by species for each rampdown event, refer to Table E4 in Appendix E.

4. Discussion

4.1. Management Question 1

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

The data collected in 2018 added another set of results for the high flow period, which started in 2016. Other than some minor differences, the results in 2018 were very consistent with those reported in the 2017 report (Sneep et al. 2018), which increases our confidence in the conclusions.

Flow releases from 2016 to 2018 were substantially higher than any other since the start of monitoring for the Lower Bridge River flow experiment. During the peak period in 2018, Terzaghi discharges completely dominated flow volumes across all of reaches 4 and 3 (7-fold higher than the Trial 2 peak), and were nearly 4-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, bank erosion and substrate deposition. 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 $m^3 \cdot s^{-1}$.

At the high flows from 2016 – 2018, the added discharge contributed additional wetted area and increased river stage by between 1.08 – 1.42 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.

During the peak period, the high flows elevated water temperatures relative to the previous flow treatments. Particularly during June, July and August, water temperatures were 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). On average, temperatures in January and February were also warmer during the Trial 3 years than the previous flow treatments. The effect was most pronounced in Reach 4, but was apparent in each of the study reaches and to some extent in the Yalakom River as well, which suggests that warmer ambient temperatures during this period in Trial 3 may have been a factor. However, assessment of mean monthly air temperatures by flow trial period for Lillooet did not corroborate this conclusion so the cause remains uncertain. Outside of these periods, the thermal regime generally 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.

For some additional information on high flow effects on physical parameters (i.e., turbidity and TDG % saturation) refer to Section 4.5.1 below.

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 1 and 2. All of the orders common in clean mountain streams were found including caddisflies (Tricoptera), Plecoptera (stoneflies), mayflies (Ephemeroptera), chironomids (midges), other true flies (Diptera), and a range of rarer taxa. All of the insects are fish food organisms (Hynes 1970, Scott and Crossman 1973, Wipfli and Baxter 2010). The mean abundances of 26,000 animals·m^{-2} (in Trial 3) to 95,000 animals·m^{-2} (in Trials 1 and 2) between trials among reaches were within ranges of densities of the same taxa found among many undisturbed rivers in North America, as reviewed in the 2017 annual report for BRGMON-1 by Sneep et al. (2018). This similarity did not change with addition of the 2018 data, and highlights that even at the reduced estimates under Trial 3, the abundance of benthic invertebrates in the LBR was still higher than the ranges described for other interior rivers in BC and Alaska (see the more fulsome description in the Year 6 (2017) report, including references (Sneep et al. 2018)). The Trial 3 abundance estimate was at the low end of the range reported for several coastal systems (e.g., Cheakamus, Capilano and Coquitlam rivers); however, the Lower Bridge River is typical of other mountain rivers with respect to the diversity and abundance of animals in the benthic invertebrate assemblage. Similarly, algal communities in the Lower Bridge River were comprised mainly of diatoms, common to periphyton in mountain rivers of temperate North America (Wehr et al. 2014).

Spatial and temporal variation among invertebrate assemblages was driven by change in abundances of chironomids, simuliids (blackfly larvae), larvae of three mayfly families

(Ephemerelidae, Heptageniidae, Baetidae), and larvae from one caddisfly family (Hydropsychidae). We limited this list of indicators to chironomids, simulids, and the mayfly indicators (Ephemerelidae plus Heptageniidae plus Baetidae), excluding the hydropsychids for reasons given in Section **Error! Reference source not found.**. This list was a subset of, and consistent with, the selection of indicator taxa found among all seasons in the 2017 analysis (Sneep et al. 2018). The indicator taxa repeatedly showed up as top discriminators of similarities of assemblages among observations within reaches and within trials and they were top discriminators of dissimilarities among assemblages between different reaches and between different trials in the 20-year compilation of 406 samples contributing to the 57 experimental units examined in the present data set. This outcome showed high precision for making management decisions using the indicators to define temporal and spatial variation among invertebrate assemblages in the Lower Bridge River according to some action, like flow release from the dam. Furthermore, all were fish food organisms, which means that change in abundance of these taxa potentially affected availability of food for fish. These changes differed by place in the river and by flow trial (Figure 3.8).

Temporal and spatial variation in invertebrate assemblages in the fall were driven by several flow variables and temperature with modification by DIN concentration driving periphyton biomass (Figure 3.10). In Trial 0, Reach 3 incubation flows were lowest on record and 16 times lower than in Reach 2 that was influenced from the Yalakom inflow. Mean temperature differed in the two reaches by less than 1°C, which likely was not enough to strongly influence invertebrate growth.

Trial 0 conditions summary:

Habitat in **Reach 3** (very low incubation flow, moderate temperature) favoured **<u>high invertebrate densities</u>** in Reach 3 (mean of 113,500 animals·m^{-2}) and **<u>high diversity</u>**.

In **Reach 2**, the habitat conditions (moderate incubation flow, moderate temperature) favoured **<u>low invertebrate density</u>** (mean of 22,000 animals·m^{-2}) but **<u>high diversity</u>**.

→ Moderate conditions for fish food organisms

In Trial 1 with a mean annual flow release of 3.1 m^3·s^{-1} (Table 1.1), Reach 4 and Reach 3 mean water temperature was highest on record (9.6 - 10.8°C) due to the release of warmer water from Carpenter Reservoir and little inflow from tributaries in those reaches. Incubation flows during sampling were low (2.5 – 2.9 m^3·s^{-1}) and there was little disturbance flow in the preceding spring and summer with a peak of only 6 m^3·s^{-1}. In Reach 2 during Trial 1, mean water temperature was lower than upstream due to the cool inflow from the Yalakom River and cooling by flow through the Bridge River canyon. Incubation flow was about double that upstream. Peak disturbance flow jumped up to 26 $m^3 \cdot s^{-1}$ associated with the Yalakom inflows and DIN concentrations increased by about 50% from those upstream potentially showing reduced nutrient deficiency of periphyton accrual in Reach 2.

Trial 1 conditions summary:

Habitat in **Reaches 4 and 3** (high temperature, low DIN, low incubation flow, low disturbance flow) favoured **moderate invertebrate densities** (mean of 68,000 – 77,000 animals·m^{-2}) but **low diversity in Reach 4 and high diversity in Reach 3**.

In **Reach 2**, the habitat conditions (lower temperature than upstream, high DIN, high incubation flow, moderate disturbance flow) favoured the **highest invertebrate densities on record** (126,000 animals·m^{-2}) and **high diversity**.

→ Good conditions for fish food organisms

In Trial 2 with a mean annual flow release from the dam of 6.1 $m^3 \cdot s^{-1}$ (Table 1.1), Reach 4 and 3 mean water temperature was high (8.2 – 10.1°C), again due to the release of warmer water from Carpenter Reservoir. Incubation flows during sampling were low (1.5 – 1.9 $m^3 \cdot s^{-1}$) and there was moderate disturbance flow in the preceding spring and summer with a peak flow of 19 $m^3 \cdot s^{-1}$. DIN concentrations in Reach 4 supported high algal PB. In Reach 2 during Trial 2, mean water temperature was again lower than upstream due to the cool inflow from the Yalakom River and cooling by flow through the Bridge River canyon. Incubation flow was more than double that upstream. Average peak disturbance flow was 45 $m^3 \cdot s^{-1}$ and mean DIN concentrations were the highest among all trial and reach combinations at 151 $\mu g \cdot L^{-1}$ supporting high algal PB. These observations show potentially less nutrient deficiency of biological production during Trial 2 than in the other trials.

Trial 2 conditions summary:

Habitat in **Reach 4** (high temperature, high algal PB, low incubation flow, moderate disturbance flow) favoured **high invertebrate densities** (mean of 88,000 animals·m^{-2}) and **low diversity**. **Reach 3** conditions (high temperature, high DIN driving high PB, low incubation flow, moderate disturbance flow) favoured **high invertebrate densities** (mean of 72,000 animals·m^{-2}) and **high diversity**. In **Reach 2**, the habitat conditions (lower temperature than upstream, high DIN driving high PB, high incubation flow, moderate disturbance flow) favoured **high invertebrate densities** (103,500 animals·m^{-2}) and **high diversity**.

Good conditions for fish food organisms

In Trial 3 with a mean annual flow release from the dam of 19.5 m^3·s^{-1} (Table 1.1), Reach 4 and 3 mean water temperature was high (8.3 – 10.2°C), again due to the release of warmer water from Carpenter Reservoir. Incubation flows during sampling were low (1.6 – 2.0 m^3·s^{-1}) and there was high disturbance flow in the preceding spring and summer with an average peak flow among years of 109 m^3·s^{-1} in Reach 4 and 111 m^3·s^{-1} in Reach 3. DIN concentrations in Reaches 4 and 3 supported high algal PB (9.7 – 10.5 µg chlorophyll a·cm^{-2}). In Reach 2 during Trial 3, mean water temperature was again lower than upstream due to the cool inflow from the Yalakom River and cooling by flow through the Bridge River canyon. Incubation flow was highest on record among all trials and reaches. Average peak disturbance flow was 130 m^3·s^{-1}. Mean DIN concentrations were moderate among all trial and reach combinations, supporting moderate mean algal PB of 5.7 µg chlorophyll a·cm^{-2}.

Trial 3 conditions summary:

Habitat in **Reach 4** (high temperature, high algal PB, low incubation flow, high disturbance flow) favoured **high invertebrate densities** (mean of 76,000 animals·m^{-2}) and **low diversity**. **Reach 3** conditions (high temperature, high algal PB, low incubation flow, high disturbance flow) favoured **low invertebrate densities** (mean of 28,000 animals·m^{-2}) and **moderate diversity**. In **Reach 2**, the habitat (low temperature, moderate algal PB, high incubation flow, high disturbance flow) favoured **low invertebrate densities** (21,000 animals·m^{-2}) and **moderate diversity**.

Poor conditions for fish food organisms

In the 2017 analysis (Sneep et al. 2018), we argued that invertebrate recruitment to the Lower Bridge River contributed to upstream to downstream change among invertebrate assemblages. We hypothesized that the dam and upstream reservoir blocked recruitment but distance from the dam resulted in increased upstream to downstream diversity and abundance. Preparation of the data for the 2018 RDA included a test for co-linearity of predictor (habitat) variables using a variance inflation factor (VIF: https://blog.minitab.com/blog/starting-out-with-statistical-software/what-in-the-world-is-a-vif). It measures how much the variance of an estimated regression coefficient increases if the predictors are correlated. It showed co-linearity between temperature and distance metrics, but we opted to include both sets of variables (temperature and distance metrics) as part of exploring wide ranging relationships among all seasons despite the statistical shortcomings. For the analyses in this report, we decided on a more stringent selection of variables using a correlation matrix with a firm cut off of 60% to select clear and independent driver variables. This latter approach resulted in selecting temperature as a potential predictor and omitting the distance metrics. Because these habitat variables were strongly correlated, temperature that was found to be an important predictor in the RDA could just as easily have been a distance metric. Both can be important (temperature affects metabolic growth and reproduction, distance may affect recruitment) and both may be contributing to what is shown only as a temperature effect in the RDA in explaining upstream to downstream change among the invertebrate assemblages.

Another way to examine the potential importance of recruitment is to compare densities and diversity of benthos in the Lower Bridge River with that in other rivers. We showed in our 2017 analysis (Sneep et al. 2018) that invertebrate densities in the Bridge River, including Reach 4, were well within those of pristine rivers not affected by impoundments. If poor recruitment was limiting density in the Lower Bridge River, we would not expect to see these similarities, especially in Reach 4 that was less than a kilometer from its source at the dam and theoretically subject to poor recruitment. Note that a reservoir blocks recruitment by drift of stream invertebrates because of little to no survival in lacustrine habitat of a large reservoir when drifting from upstream. We would expect to see much lower abundances and lower diversity compared to what was observed in the 20-year data record. Diversity measured as family richness was consistently lower in Reach 4 than further downstream (Figure 3.9), which does infer headwater type characteristics of that reach of some recruitment of taxa but not all that occurs with increasing distance from source. Density, however, was moderate to high relative to that in the other reaches, which shows that recruitment by colonizers was rapid within a distance as short as a kilometer from the dam. This evidence of rapid colonization that produced densities common to undisturbed rivers within a very short distance downstream of the dam suggests that food supply for fish is amply produced even in close proximity to the dam. Invertebrate recruitment seems less of a factor limiting food for fish when examined this way. If this rationale is correct, our selection of temperature over distance metrics in the redundancy analysis to explain change in assemblage patterns is reasonable.

The striking effect of Trial 3 was the 73% decline in mean invertebrate abundances. The redundancy analysis showed that peak disturbance flow occurring during the spring to summer flow release from the dam was the most important factor contributing to the change. All indicator taxa responded in common with flood events in streams causing scour and physical movement of particles including invertebrates due to shear forces at high flow (Robinson et al. 2004). The contrast of this response to that of earlier trials (mean annual flows of 3 $m^3 \cdot s^{-1}$ (Trial 1) and 6 $m^3 \cdot s^{-1}$ (Trial 2)) showed that an optimum flow for sustaining densities of benthic invertebrates that are fish food organisms was exceeded during Trial 3.

This Trial 3 effect was measured several weeks 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 rapid recovery. 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 the hyporrheic zone, 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).

Weak re-colonization following the flow disturbance in the Lower Bridge River suggests that one or more of these factors was limited. We argued in the 2017 report that slow recruitment was the most plausible explanation for the low density and biomass following the Trial 3 flows

(Sneep et al. 2018). Although we have now shown that benthos density and diversity was well developed within a short distance downstream of the dam, at least among the earlier trials (Trial 0, 1, 2), those data don't show what time is needed for communities to get to that point following a disturbance. Rate of recruitment can still be an issue here. In the fall during falling air temperatures, oviposition can be low and, in the absence of drift from upstream sources of recruitment, will be smaller than at other times of the year, despite optimum temperature in the river caused by warm water released from the dam and relatively high algal biomass that would supply ample food for benthos.

Colonization of Reach 4 by periphyton and invertebrates following initiation of the continuous flow release at the start of Trial 1 (after 37 years without continuous flow) occurred rather quickly (i.e., within a month of flow restoration in August 2000) (Decker, Bradford & Higgins 2008). However, in that case, invertebrate densities in the next downstream reach (Reach 3), which had been continuously wetted by groundwater and tributary inflows, were very high (>100,000 organisms/m^2 – see Figure 3.5). This would have provided an ample source for supporting rapid colonization of Reach 4. Our conclusion remains that low rate of recruitment after the flow disturbances among the Trial 3 years (due to lack of supply from upstream sources *and* reduced abundance in downstream reaches) may be important in limiting or slowing recovery of the benthic community following high peak flows approaching and exceeding 100 $m^3 \cdot s^{-1}$ relative to the conditions at the start of Trial 1.

The redundancy analysis showed the importance of eight habitat variables explaining patterns in the invertebrate assemblages. The included variables on two of eight possible axes explained 71% of the fitted variance. This value was sufficiently high that we have confidence that the two redundancy axes were a good representation of the underlying model. A note of caution is that the axes and thus the model only explained 34% of total variance. This outcome means that much of the variance was explained by factors not included in the model. We don't know what variables were missing. They may range from unmeasurable error anywhere from field measurements to the labs to actual missing and measurable variables. We do know that the variables included in the RDA model were important and significant predictors, which means that the large drivers being peak flow, temperature, incubation flow, etc. were definite factors driving assemblage patterns and can be used to draw conclusions as outlined above. Others remain outstanding within the present scope of analysis.

4.3. Management Question 3

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

The 2018 fish sampling data added another set of results for the high flow years, which started in 2016. As noted in the response to management question 1 above, other than some minor differences, the results in 2018 were very consistent with those reported in the 2017 report (Sneep et al. 2018), which increases our confidence in the conclusions.

Total juvenile abundance for the 3 study reaches was approx. 63,000 fish in 2018, which was lower than the 2016 and 2017 estimates (79,000 and 73,000, respectively). This decrease in 2018 was primarily due to reduced recruitment of mykiss fry (i.e., by approx. 6,000 – 9,000 fish relative to 2016 and 2017). Abundances of mykiss parr, coho fry and chinook fry were each similar to the estimates from the previous high flow years. The higher total estimate in 2016 was largely due to higher abundance of mykiss parr that year, which had recruited as Age-0+ fish under the final year of the Trial 2 flows in 2015. By reach, highest abundances for both mykiss and chinook juveniles were in Reach 3, followed closely by Reach 2. Coho production was fairly equivalent (and very low) between reaches 3 and 2. The most dramatic reductions in abundance, by reach, for all species and age classes was in Reach 4 across each high flow year.

Overall, juvenile salmonid abundance and biomass have been substantially reduced under the three 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 ~69,000 fish, respectively). Overall, the recruitment of juvenile salmonids was reduced by 80% under High flows (from 2016 to 2018) relative to trials 1 and 2, when production was greatest overall in each reach.

While all species and age classes have declined, the degree of effect varied among some of them. Under the high flows, the average production of mykiss fry was 20% 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. Coho fry have gone from being the second most abundant species-age class, to the lowest under the high flows. This could be due to the coincidence of the onset of high flows in May shortly after their emergence time in March or April (predicted) when their capacity to hold or select habitats in the high flows would be very limited. This same factor would also likely be an issue for the mykiss fry, which would likely emerge during the high flow period (June – July). Trends in biomass among flow treatments for all species and age classes closely matched those based on abundance because changes in average weight across flow treatments (see more on this in the final paragraph of this section) have been less significant than the changes in abundance.

While the duration and magnitude of the high flows were somewhat different in 2016, 2017 and 2018, the resulting abundance and biomass estimates among those three years were fairly equivalent (relative to changes among trials), 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 2018 did not result in substantial differences in the recruitment of fry between those years – all were equivalently low. This further supports, as was suggested in the 2017 report, 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.

Some initial (pilot) work was done in 2018 to begin exploring where on the high flow hydrograph that threshold may occur. Based on the results of this first year of sampling, the numbers of fish at a set of selected mainstem habitats appeared to decline as flow releases increased between 28 and 70 $m^3 \cdot s^{-1}$, and then remained low as flows peaked at 102 $m^3 \cdot s^{-1}$, and then were ramped back down across the month of July. One exception was for mykiss fry, which became more abundant at the selected sites on the descending limb of the hydrograph, between flows of 67 and 27 $m^3 \cdot s^{-1}$. However, this probably has more to do with the timing of their emergence which likely occurs in July as the flows were ramping down.

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).

The addition of the 2018 data points for coho and chinook (i.e., 2017 spawners vs. 2018 juvenile recruits) did not change the curves substantially because both the escapements and resulting fry recruitments were very similar to the previous high flow years for both species. Thus, more data (at different levels of escapement) are required to better clarify the initial slope of the stock-recruitment relationships which would strengthen inferences about whether spawning stock size has limited chinook and coho recruitment during any of the monitoring years. Nonetheless, these data are useful for understanding the differences in productive capacity (asymptote of each curve) of the study area for each flow treatment, which provides the same conclusions as comparison of the mean juvenile abundances across trials.

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 were almost always highest (or among the highest) in each reach during the high flow period (2016 – 2018) 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 some cases.

There are a few reasons why the mean sizes tended to be highest during the high flow years: 1) despite significantly reduced abundance of benthic invertebrates (see discussion of benthic invertebrate results in Section 4.2, above), the amount of forage available may still have been ample given the significantly reduced density of juvenile fish from 2016 to 2018 (significantly reduced fish numbers means significantly lower competition for the food resources that are available); 2) water temperatures were warmer during the spring and summer rearing period in Trial 3, which may have improved growth conditions; and, 3) the high flows likely selected for the largest individuals, as fish compete for habitat areas that are available and the smallest individuals may more likely be displaced downstream or out of the study area.

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 2018 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; McHugh and Soverel 2017; Sneep et al. 2018). Ramping across this range in 2018 generally conformed to the timing and shape implemented under the previous trial flows (see Section 3.2.3). 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 and 4.0 cm/hr in 2018) to reduce flows more quickly without increasing fish stranding risk significantly (based on results for 2016 – 2018).

An additional comment from field crews in 2018 was related to waiting for strand-risky habitats to isolate from the main channel flow before salvaging fish (so that salvage numbers reflect actual stranded fish rather than stranded fish + incidental catches). The crews noted that on ramping days when air temperatures were high, water temperatures in the strand-risky habitats were elevating quickly and causing increased stress on the fish or mortality. In an effort to minimize this stress and mortality, crews opted to capture fish as incidental catches and move them to the main channel flow sooner, rather than waiting for the habitats to isolate.

4.5. Modified Operations (High Flow) Management Questions

4.5.1. High Flow Monitoring

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

The high flows in 2018 resulted in elevated water temperatures, turbidity and %TGP levels (relative to background) in reaches 3 and 4, and caused some erosion along the wetted edge at various sites that were monitored in reaches 2 – 4.

Air and water temperatures generally increased in the LBR across the monitoring dates (as per seasonal), and were relatively consistent among locations across reaches 3 and 4 on each survey date. As noted in response to the management question in Section 4.1, the upper range of these temperatures was 3-4°C warmer than daily average temperatures for this period during trials 1 and 2. However, these temperatures were within the optimal range for rearing by chinook, coho and steelhead, as cited in the literature, which may have one of the factors contributing to the larger mean size of the juveniles from these species by the time of the annual stock assessment sampling in September (McCullough et al. 2001; Oliver & Fidler 2001; Myrick & Cech 2000).

Based on spot monitoring during the high flow period in 2018, turbidities increased with the rising limb of the hydrograph and peaked at between 36 and 38 NTU (according to location) in the second week of May, which was between 12 and 20 NTU lower than peak values measured in 2017 and about a month earlier. Turbidities then steadily diminished to between 4 and 8 NTU across the remainder of the high flow monitoring period. Turbidity values in 2018 were elevated above levels recorded at the Trial 2 peak discharge (i.e., 15 $m^3 \cdot s^{-1}$) for a total of 15 days (i.e., from 9 to 24 May).

Some of this turbidity is inherent in the water drawn from the bottom of Carpenter Reservoir; however, much of it was also due to flooding of edge areas as the flows increased from base levels. The lower turbidities in 2018 (relative to 2017) may be because the flow magnitudes were within the range of releases delivered in the previous two years (i.e., within the range of stage elevations that had already been disturbed by past flows). For reference, the peak turbidity measurement in the Yalakom River was 18 NTU (at a discharge of 18 $m^3 \cdot s^{-1}$) in the third week of June and then steadily declined to 2 NTU by the end of July. However, turbidities were not measured at the earlier peak flow (27 $m^3 \cdot s^{-1}$) on the Yalakom, which occurred in mid-May 2018.

Total dissolved gas (TDG) levels generally increased from 101% to 110 – 112% (at the plunge pool) as flows were initially ramped up between 11 and 24 May (from 15 to 70 $m^3 \cdot s^{-1}$), and then stabilized across the remainder of the high flow period. TDG levels at the two downstream sites in Reach 3 (Russel Springs and Upstream of the Yalakom) remained below 110% for the duration of the survey period. To-date, the mechanism causing the increased TDG at high flows from the low-level outlet at Terzaghi Dam has not been determined, but the observed saturation levels have been below the thresholds for triggering a mitigation response according to Table 1 of BC Hydro's Total Dissolved Gas Management Strategy (i.e., BC Hydro 2014).

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

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

Bank erosion and substrate recruitment was documented to occur at 12 of 19 monitored sites in reaches 2, 3 and 4 during 2018, covering a total area of ~5,400 m^2 (est.). There were two large areas (>1000 m^2) in Reach 2 ("Below Horseshoe" and "Horseshoe Bend"), and the remainder were considered medium- (*n*= 2) or small-sized (*n*= 8) sites in reaches 3 and 4. The other seven assessed sites did not show evidence of bank erosion or deposition under the 2018 high flows.

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

Beyond the information from the kokanee entrainment surveys (0 injured fish or mortalities observed in 2018), we don't have any data to suggest a direct adverse effect of the measured temperatures, turbidity levels, %TDG saturation or bank erosion across the high flow period on fish in the Lower Bridge River – especially relative to the effects of release temperatures during the incubation period and the effects of the high flows on rearing habitat availability, which we are more certain about.

4.5.2. High Flow Ramp Down Monitoring and Stranding Risk Assessment

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

According to the fish salvage results for ramp downs from high flows (>15 $m^3 \cdot s^{-1}$) in 2016, 2017 and 2018, the fish stranding risk was consistently low (or occasionally moderate), per 1 $m^3 \cdot s^{-1}$ increment of flow change, above a threshold of ~13 $m^3 \cdot s^{-1}$ based on the criteria defined in the fish stranding protocol (Sneep 2016). Conversely, below the 13 $m^3 \cdot s^{-1}$ threshold, the fish stranding risk was more consistently moderate or high. 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 – 2018 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 – 2018 discharges occur. For these reasons, the incidence of fish stranding and the effects of faster ramp rates on stranding risk should continue to be monitored for flows >15 $m^3 \cdot s^{-1}$ in order to build up a larger sample size of data and improve confidence in the results.

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

Under 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 were significant among them: On average, the number of stranded fish in Reach 4 (mean = ~3,000) was nearly 1.5-fold higher than the number in Reach 3 (mean = ~2,000), and the amount of identified stranding area was nearly equivalent among them (4,865 and 4,540 m^2, respectively; Sneep 2016) despite the fact that Reach 3 is nearly four times longer than Reach 4 (~12 km vs ~3 km, respectively). 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 highest in Reach 3 (8 fish per 100 m^2), followed by Reach 4 (3 fish per 100 m^2), and then reaches 2 and 1 (2 and 1 fish per 100 m^2, respectively). At high flows, total amount of identified stranding area also varied among the reaches: 4,887, 9,105, 22,900, and 30,000 m^2 in reaches 4, 3, 2, and 1, respectively.

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 – 2018 overall, it would be worthwhile to characterize the relative stranding risk among the reaches at different high flow magnitudes (when fish abundance may be greater). However, based on assessment of stage changes in Reach 2 within the Trial 2 range (see Table E5 in Appendix E), total daily stage changes per event in that reach were approx. ⅓ to ½ the magnitude of changes at the top of Reach 3, and hourly changes were likely lower as well due to the mitigating influence of the Yalakom River and other tributary inflows. This is one of the primary reasons that fish salvage efforts were focussed on reaches 4 and 3 and not on reaches 2 and 1 during implementation of the Trial 1 and 2 hydrographs, as well as flow ramp downs within that range in recent years. Considered together: the reduced stage changes, moderated ramping rate due to attenuated inflows, and generally low fish stranding risk documented for reaches 2 and 1 to-date, mutually support that fish stranding risk below the Yalakom confluence is lower than it is in the reaches above.

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

Ramping rates implemented in 2018 were between 1.2 and 4.0 cm/hr (mean stage reduction per hour at the 36.8 km compliance location). This represented the second year that rates above the ≤2.5 cm/hr WUP-referenced rate were specifically targeted (rates in 2017 were up to 4.1 cm/hr). As before, fish salvage crews were on the ground to monitor the results, but generally avoided proactively moving fish out of strand-risky habitats in advance of isolation or dewatering (i.e., "incidental" catches) such that catch data would better reflect actual numbers of stranded fish[1]. Despite the variation in ramping rates across the high flow range, the incidence of fish stranding did not change substantively relative to the identified risk for flows ≤15 $m^3 \cdot s^{-1}$. The observed stranding risk remained low (<10 per 1 $m^3 \cdot s^{-1}$) to moderate (10 to 99 fish per 1 $m^3 \cdot s^{-1}$), as defined in the fish stranding protocol (Sneep 2016), across each of the implemented rates at high flows.

Currently the sample size for stranding monitoring at ramping rates >2.5 cm/hr is still small. As was noted for the MQ above, juvenile fish abundances in 2017 and 2018 were low overall, which could have confounded the incidence of stranding despite the higher rates in both of those years. However, the results to-date suggest that stranding risk is lower at flow releases >13 $m^3 \cdot s^{-1}$ (see above). As such, this should provide opportunity to further test higher rates across the high flow range going forward without unduly risking higher fish mortality. Increasing the number of ramp down events completed at higher ramp rates will be necessary to reduce uncertainty about the specific effects of higher ramp rates across the different high flow levels.

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

With the inclusion of the 2018 site reconnaissance and salvage survey data, the distribution of sites was near equal at 45% on river left and 55% on river right across the high flow range (>15 $m^3 \cdot s^{-1}$); whereas, within the previous trial flow range (≤15 $m^3 \cdot s^{-1}$), the distribution was 80% and 20%, respectively. Upon initial purview, differences in distribution of sites according to side

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

of the river may seem unexpected, given that there is no known reason based in an understanding of channel morphological processes that more strand-risky habitats would naturally form on one side of the river versus the other across the length of these reaches. Rather, it's possible the reason could have more to do with human-caused effects than natural ones.

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

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

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

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

In some cases, such as in the past 3 high flow years, there can be additional rationale for ramping the flows down faster in order to reach more optimal summer rearing flows (i.e., ≤15 $m^3 \cdot s^{-1}$) more quickly following peak flows. With the data for high flows available from 2016 to 2018, there is some evidence for when faster ramping rates can be applied without unduly increasing fish stranding risk. As described in the management question responses above, this could apply to ramping rates up to 4 cm/hr at discharges >13 $m^3{\cdot}s^{-1}$ based on the information currently available. However, due to the factors noted in the sections above (low fish abundance during the high flow years to-date; low sample size at higher ramping rates), the application of these rates should be accompanied by ramp monitoring and fish salvaging (as was done in 2016 – 2018) to further flesh-out the fish stranding risk at flows >15 $m^3{\cdot}s^{-1}$ when fish abundances may be greater, and expand the dataset from which conclusions are drawn.

4.5.3. Juvenile Salmonid Habitat Availability and Displacement

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

Based on the results of the pilot-level sampling in 2018, there is insufficient information at this point to answer this management question (i.e., from the field data alone). Depths and velocities were highly variable across the different mainstem sites (both those pre-selected as high quality habitat and low quality habitat), and tended to be more consistent in the off-channel sites which are more sheltered from the main channel flows. The high quality mainstem sites tended to span a slightly broader range of both depth and velocity across the high flows than the low quality sites, suggesting they were somewhat more directly affected by changes in discharge across the high flow period. There was some evidence that changes in the availability of suitable habitat areas and conditions (i.e., depths & velocities) at some locations were offset by the changes at other sites, such that there was no substantial net loss among the pre-selected sites across the high flow range surveyed in 2018. However, this is highly tenuous given the degree of variability in the data and the limited sample size (i.e., number of sites and flows tested).

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

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

For the fry age class of coho and chinook, the densities in the pre-selected mainstem sites appeared to diminish at flows between 28 and 70 $m^3{\cdot}s^{-1}$ on the ascending limb of the hydrograph in 2018, and this was true in both high and low quality sites. Mykiss fry abundance was low across the ascending limb and peak portions of the high flow hydrograph, and then densities at the mainstem sites increased in the latter part of the descending limb (i.e., when flows were between 82 and 27 $m^3 \cdot s^{-1}$, depending on the site). This likely had to do with emergence timing for the new year-class of mykiss, which coincided with the timing of this portion of the hydrograph in 2018. Mykiss parr densities tended to be highest on the ascending limb of the hydrograph (i.e., up to 70 $m^3 \cdot s^{-1}$), and they appeared more tolerant of high flows than the coho and chinook fry. This is likely due to the bigger body size and better swimming capability of the parr lifestage, which allows them to exploit a broader range of depths and velocities and move among locations or habitats as conditions change.

There was potentially a slight increase in coho fry abundance in the Applesprings off-channel site on the descending limb of the hydrograph, and slightly higher densities of mykiss parr in both off-channel habitats on the peak flow survey date. But beyond this, there were no substantial changes in density in the off-channel sites that would point to significant immigration or emigration between these habitats and the mainstem during the high flow period in 2018, particularly for fry.

These data provide some preliminary insights into potential flow thresholds affecting juvenile salmonid rearing habitat use in the Lower Bridge River mainstem above the Trial 1 and 2 hydrographs. However, any conclusions from these results are tenuous at this point due to a number of factors: 1) lack of information on capture probability and how it varied by species, age class, site and flow rate across the high flow range; 2) limited number of flow intervals tested on both the ascending and descending limbs of the hydrograph for defining potential flow thresholds more specifically; and 3) uncertainty about how the patterns described for the selected sites applies at the reach-level or within the study area as a whole.

4.5.4. Substrate Mobilization, Deposition and Composition Monitoring

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

There were no data available within the scope of the BRGMON-1 program in 2018 for answering this question. Refer to the Year 6 report (Sneep et al. 2018) and the latest report by KWL on their substrate mobilization, deposition and composition monitoring (Ellis et al. 2018) for some relevant information pertaining to this question. We understand that work is also being conducted by BC Hydro using the Telemac2D model to make predictions about the amount and distribution of rearing habitat area for the various salmonid species across the range of high flows.

5. Recommendations

In addition to the six recommendations provided in the Year 6 (2017) report, the following recommendations stem from the analysis and reporting of results for addressing the management questions up to, and including, Year 7 (2018):

1. In order to better address the Modified Operations Management Question #4: "How does juvenile salmonid habitat availability in the Lower Bridge River change with discharge under the modified flow regime?", this program could utilize outputs from BC Hydro's Telemac2D model for assessing the predicted availability of suitable rearing habitats for juvenile chinook, coho and mykiss in the Lower Bridge River across a range of high flows. This effort should link together the information provided by the model with the habitat suitability curves and the data that have been collected in the field as a part of the Juvenile Salmonid Habitat Availability and Displacement task. Integrating the information provided by the model into the BRGMON-1 reporting for addressing this management question will require discussion between the biologist in charge of the analysis and reporting, and relevant staff from SER and BC Hydro in order to ensure that all of this information can be synthesized appropriately and effectively.
2. In subsequent high flow years, continue the juvenile salmonid habitat availability and displacement sampling that was piloted in 2018. Given the variability associated with the data analyzed for the set of pre-selected sites to-date, consider increasing the number of mainstem sites (e.g., up to 8 each in reaches 3 and 4) and the number of flow steps surveyed (e.g., every 10–20 $m^3 \cdot s^{-1}$ on the ascending and descending limb of the hydrograph) to improve confidence in the results and provide better resolution about what flow ranges fish displacement increases for informing flow management decisions. Methods for quantifying capture probabilities among sites, survey dates and flow release magnitudes (e.g., mark-recapture – see methods in Korman et al. 2010) should be considered in order to generate more reliable and comparable density estimates across the high flow range.
3. Expand monitoring to include Reach 1 as described in the Reach 1 Feasibility Assessment memo report (Sneep et al. 2019). Based on this assessment, the extension of sampling into Reach 1 would be both feasible and directly compatible with sampling and data collection that has been established in the other BRGMON-1 study reaches. Four highly suitable sampling locations were identified for this feasibility assessment. By adhering to the same methods, gear, and sampling schedule employed in reaches 2, 3 and 4, the data generated by this expanded scope could readily be run through the existing data analysis routines for assessing primary and secondary productivity (periphyton and benthos), habitat attributes that are driving this production, as well as juvenile abundance and biomass for any years and flow regimes that are monitored going forward. Understanding the contribution of Reach 1 may help establish some

additional context for the reduced production or retention of benthos and fish observed in the other study reaches under high flows in recent years and address the long-advocated desire of Xwísten to have a comparable set of results for Reach 1 to include in flow discussions and decisions for the Lower Bridge River.

4. Continue stage monitoring at the existing locations in reaches 4, 3 and 2 (as well as the new location proposed for the bottom of Reach 1 starting in 2019) to confirm the differences in stage changes associated with ramp down events among the reaches. Some additional data to support that stage changes are reduced and ramp rates are moderated below the Yalakom confluence would be beneficial for supporting current conclusions that fish stranding risk is systematically lower in reaches 2 and 1 than in reaches 4 and 3.

5. Juvenile abundance sampling (i.e., standing stock assessment) is proposed for: a) the usual 49-50 sites in reaches 4, 3 and 2; b) 8 sites in two off-channel habitats (i.e., Bluenose and Applesprings – Mitigation Effectiveness Monitoring); as well as c) 12 new sites in Reach 1 for 2019. Given this expanded scope (i.e., a new total of 69 depletion sites spanning from Terzaghi Dam to the confluence with the Fraser River), it is imperative that a sufficient number of experienced crews are employed in order to complete the larger suite of sites within the same general time period as stock assessment sampling in previous study years (i.e., within the first two-to-three weeks of September). Maintenance of the existing sampling schedule, particularly for the existing mainstem sites in reaches 2 to 4 which have the longest time series, is paramount. Where additional time may be required, we recommend sampling in reaches 2 to 4 be as consistent as possible with dates in previous years, with extra time for Reach 1 and off-channel sites completed after, as necessary.

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

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 $l_3 w_3$ 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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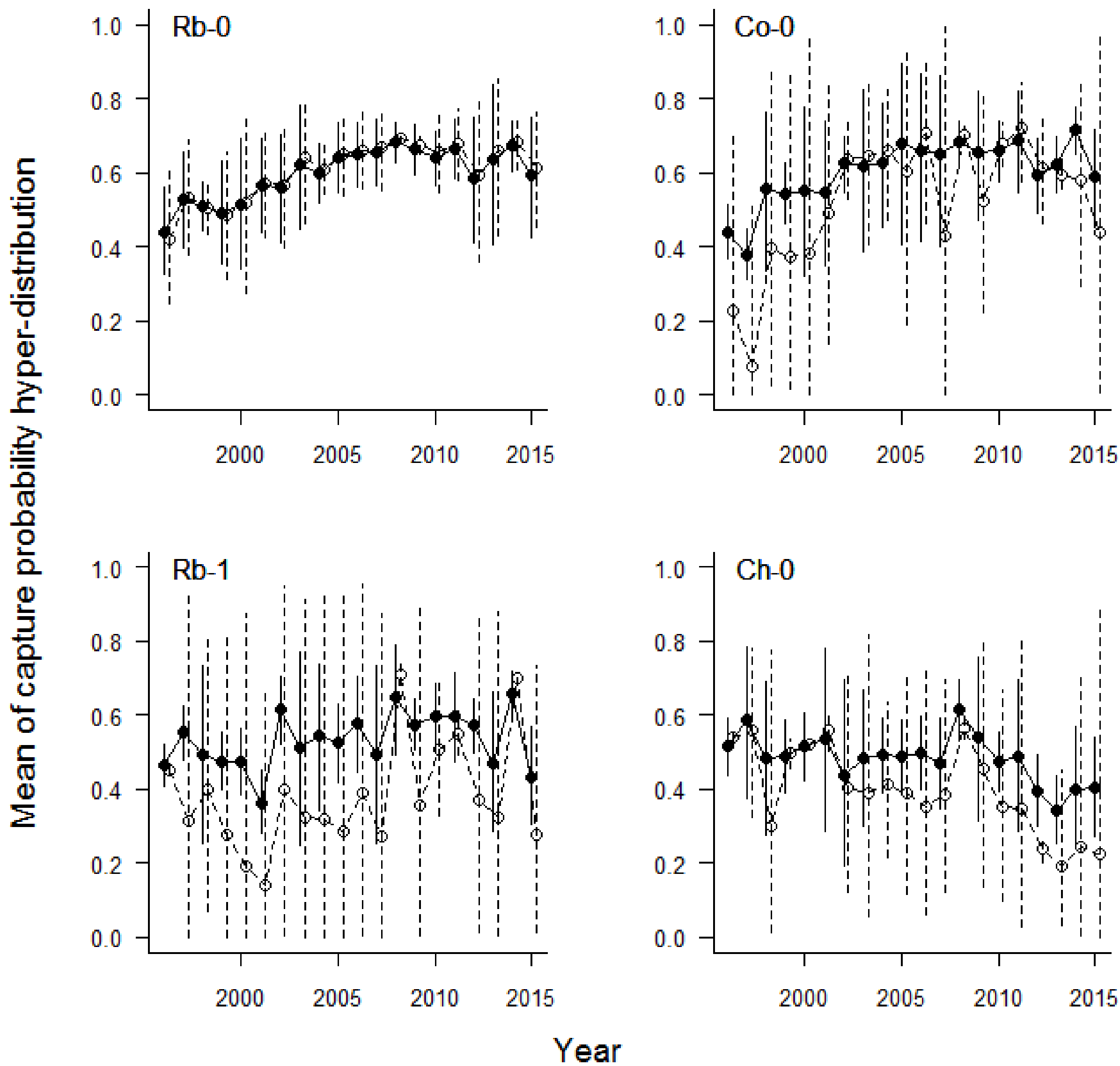


Figure A1. Annual estimates of the mean (with 90% credible interval) of the capture probability hyper-distribution (distribution of capture probability across sites) based on the HBM with more restrictive priors for the capture probability hyper-distribution (solid symbols). Also shown are estimates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

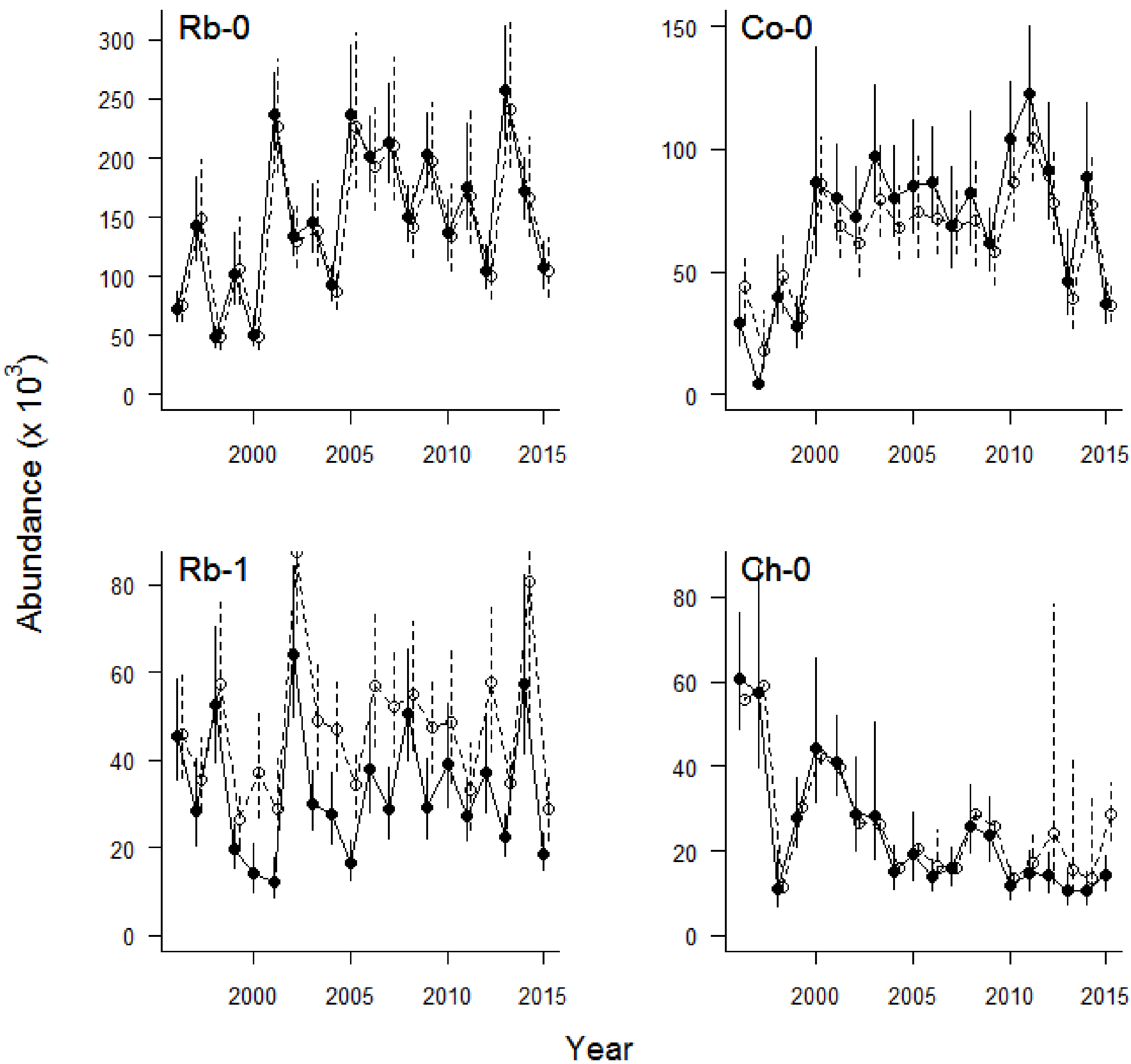
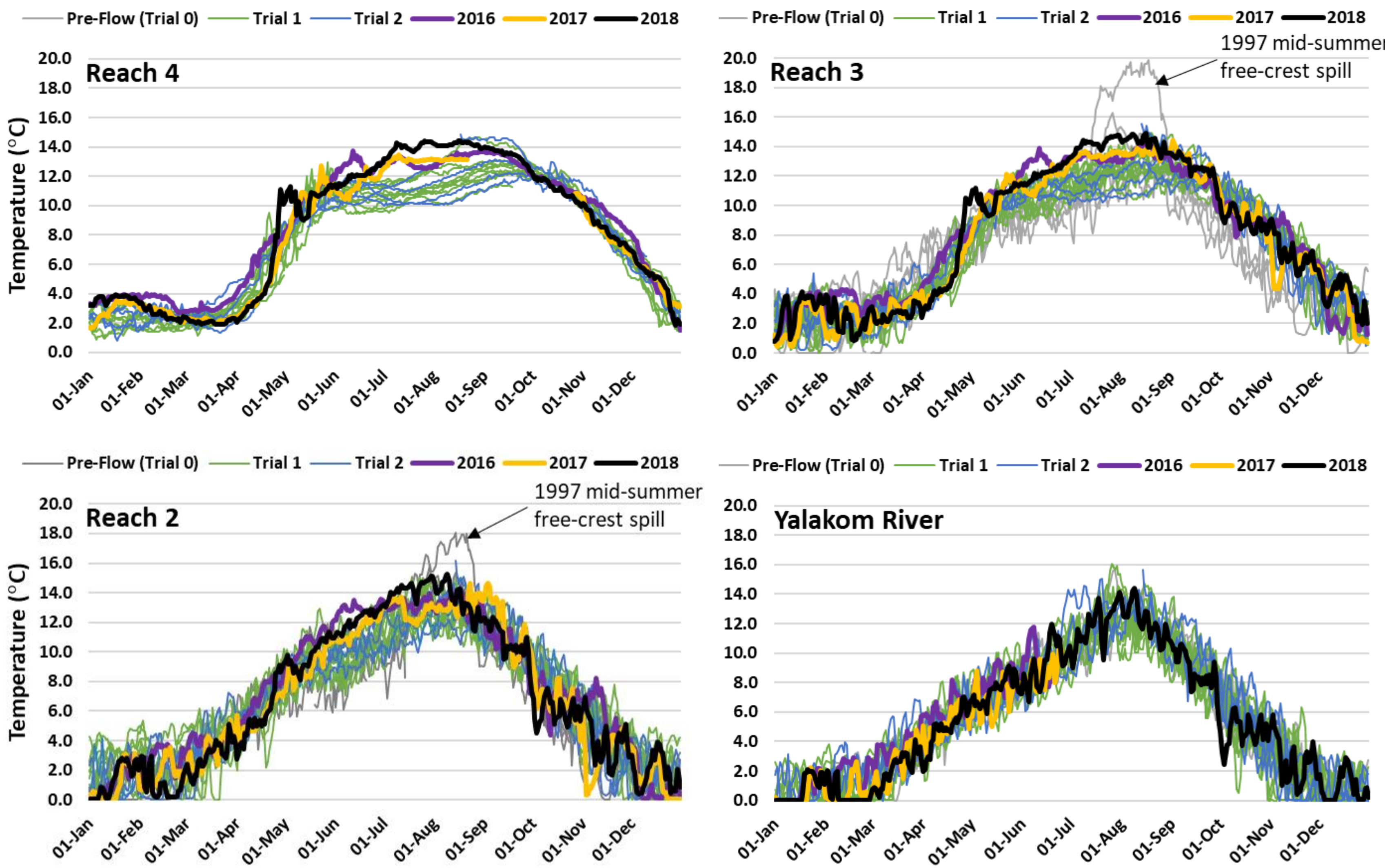
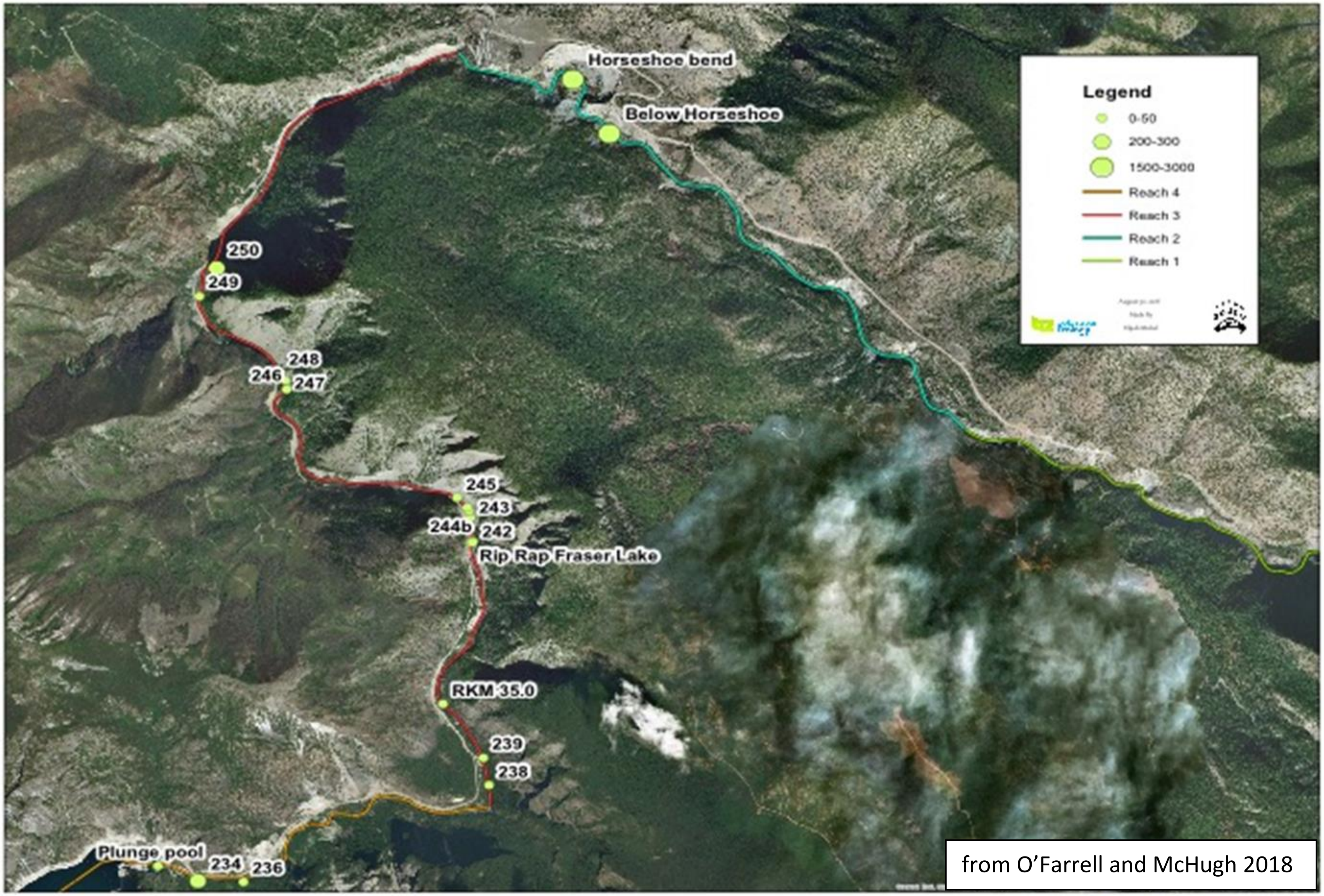


Figure A2. Annual estimates of abundance (all reaches combined) based on the HBM with a more restrictive prior (solid symbols). Also shown are estimates based on the uninformative priors used in Bradford et al. (2011, open symbols).

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

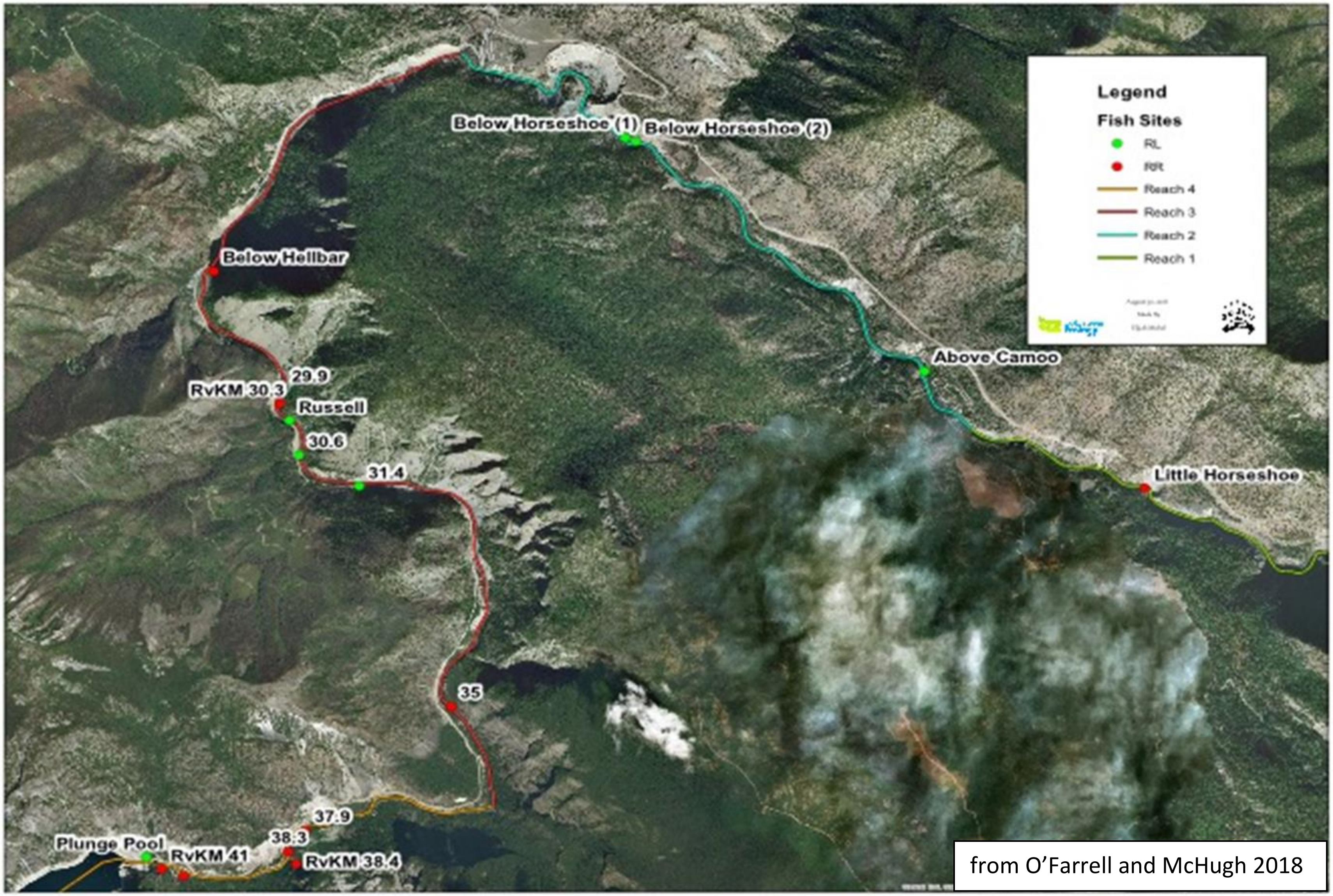


Appendix C – LBR Bank Erosion and Sediment Recruitment Sites assessed during High Flows in 2018.



from O'Farrell and McHugh 2018

Appendix D – Locations of Potential Fish Stranding Sites based on Reconnaissance Surveys at High Flows in 2018.



from O’Farrell and McHugh 2018

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

Table E1 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2018.

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

[a] These values are based on the discharge-stage relationship (see Figure 3.27 in Section 3.2.3) since stage values for the Rkm 36.8 logger were not available on these dates in 2018.

Table E2 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2017.

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

Table E3 Detailed Summary of Flow and Stage Changes, and Ramping Rates Associated with Individual Rampdown Events in 2016.

Year	Date	Event #	Ramp Duration (hours)	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Start Stage (cm)	End Stage (cm)	Stage Change (cm)	Mean 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 E4 Fish salvage tallies by species, ramp date and reach – 2018 results (from O'Farrell and McHugh 2018).

Species	Jul 4	July 10	July 17	July 18	July 24	July 25	July 31	Aug 1	August 2	August 8	August 9	August 15	August 16	August 21	August 22	October 2	October 3	TOTAL
Reaches 1, 2																		
Chinook	8	16	10	20	23	4	12	0	0	0	0	0	0	0	0	0	0	93
Coho	4	40	60	20	115	46	206	0	0	0	0	0	0	0	0	0	0	491
Steelhead/ RB	10	22	3	28	201	0	109	0	0	0	0	0	0	0	0	0	0	373
Bull trout	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	3
Mountain whitefish	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	4
Sucker spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Redside shiner	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sculpin spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sockeye/Ko kanee	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Subtotal	25	78	73	69	342	50	327	0	0	0	0	0	0	0	0	0	0	964
Reaches 3, 4																		
Chinook	0	1	2	12	2	1	0	6	0	0	88	0	28	5	2	0	7	154
Coho	3	10	6	27	30	11	23	29	14	2	48	10	10	14	7	0	9	253
Steelhead/R B	30	13	87	110	117	49	45	88	27	71	679	20	221	141	70	7	113	1888
Bull trout	0	1	0	0	0	0	0	0	0	0	4	0	0	1	0	0	1	7
Mountain whitefish	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	3
Sucker spp.	0	0	0	0	0	0	0	5	5	0	0	3	0	0	0	0	0	13
Redside shiner	0	0	0	0	0	0	0	82	61	0	0	5	0	0	0	0	0	148
Sculpin spp.	0	2	0	2	0	1	0	40	28	1	0	47	3	3	461	27	7	622
Sockeye/Ko kanee	0	0	0	0	0	0	0	0	0	0	0	9	0	0	0	0	0	9
Subtotal	33	27	95	151	149	62	68	250	135	74	822	94	262	164	540	34	137	3097

Table E5 Summary of stage changes at available monitoring locations in reaches 2, 3 and 4 for each rampdown event in 2018.

Year	Date	Event #	Start Flow ($m^3 \cdot s^{-1}$)	End Flow ($m^3 \cdot s^{-1}$)	Flow Change ($m^3 \cdot s^{-1}$)	Stage Change (cm)			
						Top of Reach 3 (Rkm 36.8)[a]	Bottom of Reach 3 (Rkm 26.0)[b]	Horseshoe Bend (Rkm 23.6)	Bottom of Reach 2 (Rkm 20.0)
2018	4 Jul	1	102.0	82.6	-19.4	-16		-6	-12
	10 Jul	2	82.9	66.9	-15.9	-18		-7	-7
	17 Jul	3	67.2	55.2	-12.0	-15		-7	-8
	18 Jul	4	55.3	44.2	-11.0	-18		-7	-8
	24 Jul	5	44.4	35.1	-9.3	-14		-5	-7
	25 Jul	6	35.1	27.1	-8.0	-14		-4	-7
	31 Jul	7	27.2	20.0	-7.2	-15		-4	-7
	1 Aug	8	20.0	15.1	-4.9	-11		-3	-5
High Flow Rampdown Summary		**8**	**102.0**	**15.1**	**-86.9**	**-123**		**-45**	**-63**
2018	2 Aug	9	15.1	11.0	-4.1	-10		-4	-7
	8 Aug	10	11.1	9.3	-1.8	-5		-2	-4
	9 Aug	11	9.3	7.7	-1.6	-5		-2	-2
	15 Aug	12	7.8	6.4	-1.3	-6		-2	-3
	16 Aug	13	6.4	5.2	-1.3	-6		-2	-2
	21 Aug	14	5.2	4.1	-1.1	-7		-2	-3
	22 Aug	15	4.1	3.0	-1.1	-8		-2	-2
	2 Oct	16	3.1	2.1	-1.0	-9		-3	-3
	3 Oct	17	2.1	1.5	-0.6	-6		-1	-2
WUP Rampdown Summary		**9**	**15.1**	**1.5**	**-13.6**	**-62**		**-22**	**-32**

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

[b] Data from this monitoring location were not available in 2018 because the logger was lost due to the high flows.