Bridge River Project Water Use Plan

Lower Bridge River Aquatic Monitoring

Implementation Year 5

Reference: BRGMON-1

2016 Annual Data Report

Study Period: January 1, 2016 – December 31, 2016

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EXECUTIVE SUMMARY

The main objectives of the Lower Bridge River Aquatic Ecosystem Monitoring program in 2016 were to: 1) reduce uncertainty regarding the effects of the flow releases on the aquatic productivity of the ecosystem; 2) inform a summer and fall rampdown strategy that reduces the risk of fish stranding while meeting environmental objectives and to salvage fish during river ramping; and 3) inform the adaptive management of the Lower Bridge River (LBR).

This monitoring program was designed to test two flow releases (Trials 1 and 2) against a zero-flow baseline scenario (Pre-Flow) released from the dam according to prescribed hydrograph shapes for each trial. The Pre-Flow release represented baseline ecological monitoring. Trial 1 was a 3 $m^3 \cdot s^{-1}$ mean annual flow (2000-2010) based on a hydrograph that ranged from a minimum of 2 $m^3 s^{-1}$ to a maximum of 5 $m^3 \cdot s^{-1}$. Trial 2 was a 6 $m^3 \cdot s^{-1}$ mean annual flow (2011-2015) that ranged from a minimum of 1.5 $m^3 \cdot s^{-1}$ to a target peak flow of 15 – 20 $m^3 \cdot s^{-1}$. In 2016 the flow release deviated from the prescribed Trial 2 hydrograph during spring and summer and a mean annual flow of 22 $m^3 \cdot s^{-1}$ was released from the dam with a peak flow of 97 $m^3 \cdot s^{-1}$.

Five monitoring activities were conducted as part of the program in 2016: 1) water temperature and stage level; 2) water chemistry, aquatic invertebrate abundance and diversity and periphyton accrual; 3) juvenile salmonid growth sampling; 4) fall standing stock assessment; and 5) habitat surveys. In addition, a rampdown monitoring component was conducted during the summer and fall seasons to minimize fish stranding risk, salvage fish and to collect information in order to inform an optimal strategy for ramping down discharge on the LBR.

Stage ranged from a low winter flow of 1.5 $m^3 \cdot s^{-1}$ and peaked at a discharge of 97 $m^3 \cdot s^{-1}$, before returning to WUP target flows in late July. Fall temperatures were on average 2°C to 4°C warmer across 2016 and both flow trial periods (Trials 1 and 2), compared to the Pre-Flow period. These effects appeared strongest in the upper reaches (reaches 3 and 4) and weakest in Reach 2. The warmer temperatures may have had implications for the life-cycle of Chinook salmon by causing early emergence and decreasing winter survival of juveniles. High flows increased the amount of wetted area in the river, but likely did not increase the amount of suitable juvenile rearing habitat across the reaches. An increase in wetted area corresponded with an increase in the cascade/ rapid habitat type throughout the study area, which may have reduced the suitability of rearing habitat as flow velocities were likely increased to above optimal thresholds. High spring and summer flows in 2016 may have displaced fish from unsuitable habitat, inhibited the accessibility of potentially useable rearing habitat, and potentially impeded successful steelhead reproduction or rearing during the spring and summer. Likely in combination with other factors unrelated to flow, the total abundance of juvenile salmonids and benthic invertebrates declined across the Lower Bridge River. Overall, while the long-term influence of high flows on fish productivity remains uncertain at this time, high flows likely impacted juvenile populations of coho and rainbow/ steelhead and contributed to a substantial reduction in aquatic productivity within the Lower Bridge River in 2016.

Limited other productivity data outside of WUP target flows were available to make definitive conclusions regarding whether high flows or low flows benefit the ecosystem more, or precisely how much influence the 2016 hydrograph may have had on the productivity of the aquatic ecosystem. More high flow data are expected in 2017 to continue to reduce uncertainty surrounding the study objectives, and further support or refute interpretations of program hypotheses.

BRGMON-1 STATUS of OBJECTIVES, MANAGEMENT QUESTIONS, and HYPOTHESES after Year 5

Study Objectives	Management Questions	Study Hypotheses	Implementation Year 5 (2016) Status
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 (Trials 1 and 2) to better inform decisions 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 of the Lower Bridge River aquatic ecosystem.	1) How does the instream flow regime alter the physical conditions in aquatic habitats of the Lower Bridge River ecosystem?	H_0: “High flow is better” H_A: “Low flow is better”	**<u>Physical Conditions:</u>** • H_0 is not rejected • H_A is not rejected **Temperature Rationale:** Flows across Trial 1, Trial 2 and 2016 appeared to cause similar temperature effects on the physical conditions of the aquatic habitat. Fall water temperatures were on average 2°C to 4°C warmer and declined more gradually than the Pre-flow period. The higher the flow release the further downstream the temperature effects extended. The Null hypothesis cannot be rejected at this stage from temperature inferences alone. We are on track to answering this management question and more data are expected under higher flows in 2017 to further reduce this uncertainty.
			Habitat Rationale: High flows increased the amount of wetted area in the river, which corresponded with an increase in the cascade/ rapid habitat type. This may have reduced the suitability of rearing habitat, as flow velocities were likely increased to above optimal thresholds. High spring and summer rearing flows in 2016 may have displaced fish from habitat and inhibited the accessibility of potentially useable rearing habitat, but rearing habitat data gaps exist at high flows. The Null hypothesis cannot be rejected at this stage as more data are required, however we are on track to answering this management question. We are expecting more data in 2017.

Study Objectives	Management Questions	Study Hypotheses	Implementation Year 5 (2016) Status
Same as above	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?	H_0: "High flow is better" H_A: "Low flow is better"	**Community Composition and Productivity of Benthic Invertebrates** • H_0 is not rejected • H_A is not rejected **Primary production Rationale:** Periphyton accrual data do not appear to be different across the flow trials. Differences in trends appear to be more closely associated with deposition and accumulation of nutrients from pink salmon spawning years in pink (odd years) and non-pink (even years) than flow regime. Null hypothesis cannot be rejected at this stage from primary productivity inferences alone. **Secondary production Rationale:** The rewetting of Reach 4 benefited the benthic invertebrate community after the initiation of flow. During Trials 1 and 2, no significant differences were observed in response to flow changes. Higher flows did not significantly increase the benthic invertebrate community abundance, or benefit the community composition. In 2016 abundance declined, while diversity remained stable. Null hypothesis cannot be rejected at this stage, and more data are needed under high flows to further test the hypothesis. .

Study Objectives	Management Questions	Study Hypotheses	Implementation Year 5 (2016) Status
Same as above	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?	H_0: "High flow is better" H_A: "Low flow is better"	**Fish:** • H_0 is not rejected • H_A is not rejected Juvenile fish production increased significantly and in Trial 1, and did not differ in Trial 2, relative to the Pre-Flow Trial. This was mainly due to the rewetting of Reach 4. Rainbow and Coho fry benefited from the flow release in both Trial 1 and Trial 2. In contrast, Chinook fry production declined across Trials. In 2016, productivity declined for rainbow fry and coho fry, while Chinook populations remained similar to previous years since the initiation of the flow release. More data under high flows are needed, and the Null hypothesis cannot be rejected at this stage. However we are on track to answering this management question.
To inform a summer and fall rampdown strategy that reduces the risk of fish stranding while meeting environmental objectives and to salvage fish during river ramping. The scope of the 2016 MON-1 monitoring of flow rampdown and fish stranding was limited to flows reductions <15 $m^3 \cdot s^{-1}$., and data summarizing rampdown results for flows > 15 $m^3 \cdot s^{-1}$, and data summarizing flow reduction monitoring > 15 $m^3 \cdot s^{-1}$ can be found in an additional 2016 LBR Spill Impact report.	4) 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 $m^3 \cdot s^{-1}$?	N/A in 2016	**Stranding Risk:** According to the BC Hydro LBR Fish Stranding Protocol, stage changes associated with the lowest fish stranding potential occurred between 15 and 9 $m^3 \cdot s^{-1}$. In 2016, data further supported these conclusions; stranding risk with the lowest stranding potential occurred between 15.3-$m^3 \cdot s^{-1}$ and 7.7 $m^3 \cdot s^{-1}$. As flows were further reduced, stranding risk increased. We are on track to answering this management question, however stranding risk may change annually with high flows and more data are needed to continue to further reduce this uncertainty. More data are also needed to reduce uncertainties regarding how ramp rates relate to stranding potential.

Table of Contents

List of Tables

List of Figures

Appendices

1.0 INTRODUCTION

The Bridge River, a tributary of the middle Fraser River, is an important fish-bearing river in Southern Interior British Columbia. While it was used historically as a major food source for St'at'imc fishing, today it is used for a variety of purposes including hydroelectric power. Traditionally, fish comprised 60% of the local diet (Kennedy and Bouchard, 1992) some of which originated in the Bridge River. However, the benefits to society from this fish resource extended much farther than just as a source of food. This fishery was also integral to a complex trading network where salmon and salmon oil were highly prized and considered the foundation of commerce in the region. The health and productivity of the Bridge River aquatic ecosystem contributed to the rich fish resource and culture in St'át'imc territory. Overall, this resource generated significant benefits towards the health and well-being of the St'át'imc Nation and trading partners.

In 1960, the Bridge River was fully impounded by Terzaghi Dam (formerly called Mission Dam), which was built at the head of a long, narrow canyon approximately 40 km upstream of the confluence with the Fraser River. This impoundment created Carpenter Reservoir, which serves as a water source for hydroelectric production in the Seton watershed, and fragmented the Bridge River, creating a controlled lower section called the Lower Bridge River (LBR). Initially, all flow was diverted to Seton Lake for hydroelectricity, with the exception of infrequent high-water spill over events. Consequently, 4kms of river directly below the dam were dewatered for 40 years (1960-2000). Downstream of the dewatered reach, and upstream of the confluence with the Yalakom River, groundwater and tributary influence created a flow less than 1% of the historic mean annual discharge (Longe and Higgins, 2002).

Concerns were raised and discussed over the lack of water flowing in the Lower Bridge River by the St'át'imc, federal and provincial regulatory agencies, and the public. After discussions in the 1980s, an agreement was reached to continuously release water to provide fish habitat downstream of Terzaghi Dam. Under the Water Use Plan, an adaptive management approach was recommended by the WUP Consultative Committee along with an environmental monitoring program, which was designed to test two main flow releases (Trials 1 and 2) against a zero-flow baseline scenario (Pre-Flow), which represented the previous 40 years. As part of a structured decision-making process (Failing et al., 2004; Failing et al., 2013) key benefits from the aquatic ecosystem were identified, and parameters were chosen and monitored during 2016 as they have been historically over the course of the Flow Trial experiment. The focus of the LBR WUP includes the physical conditions in the aquatic and riparian habitats, biomass and growth of juvenile salmonids, periphyton and benthic invertebrate abundance and diversity as a proxy for river health. This program gathers empirical data to inform the flow management of the LBR, and aims to generate a better understanding of the effects of the introduction of water from Carpenter Reservoir on the aquatic ecosystem productivity and the ecosystem services, or benefits which the river generates, below the dam.

An average 3.0 $m^3 \cdot s^{-1}$ annualized interim water budget (Trial 1), based on a hydrograph that ranged from a minimum of 2 $m^3 \cdot s^{-1}$ to a maximum 5 $m^3 \cdot s^{-1}$ was initially allocated for in-stream flow releases into the LBR. Water was released on August 1, 2000 and continued at this level until spring 2011. Prior to this release, data were collected from 1996-2000 (Pre-Flow), to provide baseline information on the pre-release ecosystem and the ecological services the river provided, and to facilitate measuring and comparing the response of the aquatic environment to different flow trials. Between 2011 and 2015 (Trial 2) the LBR annual hydrograph target was 6 $m^3 \cdot s^{-1}$, and ranged from a minimum of 1.5 $m^3 \cdot s^{-1}$ to a maximum of approximately 20 $m^3 \cdot s^{-1}$. In 2016, risk reduction measures to address seismic and seepage issues at Lajoie Dam, critical outages associated with the capital replacement plan for Bridge River 1 and 2 generating station units, and work at Seton and Lajoie generating stations affected the volume and timing of flow releases from Terzaghi Dam in 2016, differing from Water Use Plan (WUP) target flows. The 2016 hydrograph average was 22 $m^3 \cdot s^{-1}$, with a summer peak flow of 97 $m^3 \cdot s^{-1}$ and a winter low flow of 1.5 $m^3 \cdot s^{-1}$.

This report describes the results of the fifth year (2016) within a ten-year study of the LBR in accordance with the Water Use Plan (WUP) Order to release water and monitor the environmental benefits and impacts of the flow release on the aquatic ecosystem. Data from this monitoring program will be used to inform the management of the LBR flow regime. Presently, the St'át'imc Nation, the Bridge River Band, BC Hydro, regulatory agencies and other stakeholders work together to determine a long-term flow release strategy for the LBR. The implementation of this aquatic monitoring program is part of the Bridge-Seton Water Use Plan. St'át'imc Eco- Resources (SER), an incorporated company owned by the St'át'imc Chiefs Council, has been contracted by BC Hydro to undertake this work. Subsequently, Coldstream Ecology, Ltd. has been subcontracted to implement the monitoring program. Detailed descriptions of past monitoring activities and results of past years can be found in McHugh and Soverel (2013 - 2015), Riley et al. (1997, 1998), Higgins and Korman (2000), Longe and Higgins (2002), Sneep and Higgins (2003, 2004), and Sneep and Hall (2005 - 2012).

1.1 Management Questions

The goal of this ecological monitoring program is to utilize an adaptive management framework to reduce uncertainty about the expected benefits of releasing water from Carpenter Reservoir downstream of Terzaghi Dam. Past studies have been unable to provide scientifically defensible predictions of the ecological benefits of the flow releases, and this lack of certainty constitutes a major challenge for decision-making regarding valued ecological resources and energy management. Consequently, the long-term monitoring program was designed to provide defensible data defining the functional relationship between the magnitude of flow releases, and physical and biological responses in the LBR channel. As identified in the WUP Terms of Reference (BC Hydro, 2012) for this monitoring program, four key management questions that directly describe the uncertainties and the learning objectives include:

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

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

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?

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 $m^3 \cdot s^{-1}$?

Juvenile salmonid biomass is used as a primary criterion to compare performances of different flow levels because salmon represent a highly valued ecological component of the aquatic ecosystem. In addition, juvenile salmonid biomass integrates the effects of flow on trophic productivity and habitat conditions in the LBR. The monitoring program was designed to test the following hypotheses regarding the ecological benefits and the effects of flow on the fish populations in the LBR:

> H_0: "High flow is better"
> H_A: "Low flow is better"

The data provided in this annual data report summarize the 2016 program. These data are part of a larger dataset (1996-2016), which will address management questions 1-3 (above) to inform long-term water use planning in 2017 and beyond. The fourth management question (above) is being addressed by a rampdown monitoring component that was integrated into this WUP monitoring in 2012. Information collected from this component will help to mitigate the risk of fish stranding and inform the optimal "shape" of the hydrograph throughout annual rampdown activities.

1.2 Objectives and Scope

The primary objectives of this monitoring program are to: 1) to reduce uncertainty regarding the effects of the flow releases on the aquatic productivity of the ecosystem; and 2) to inform a summer and fall rampdown strategy that reduces the risk of fish stranding while meeting environmental objectives and to salvage fish during river ramping. The scope of 2016 BRGMON-1 monitoring of flow rampdown and fish stranding was limited to <15 $m^3 \cdot s^{-1}$, and data summarizing rampdown results for flows > 15 $m^3 \cdot s^{-1}$ can be found in the 2016 LBR Spill Impact report (McHugh et al, 2017). Specifically, monitoring program activities in 2016 continued to focus on:

1) Water temperature, dam discharge, and river stage;
2) Water chemistry parameters, periphyton accrual and diversity, and the relative abundance and diversity of aquatic invertebrates during the fall field ecology series;
3) Growth, distribution, and relative abundance of juvenile salmonids including coho salmon (*Oncorhynchus kisutch*), chinook salmon (*O. tshawytscha*), steelhead and rainbow trout (*O. mykiss*), within the study area;
4) Summer and fall rampdown monitoring and salvage activities; and
5) Aquatic habitat assessment.

1.3 Study Area

The Bridge River lies within St'át'imc Territory, in Southern Interior British Columbia. The Lower Bridge River is the section between the confluence of the Fraser River and Terzaghi Dam. It is divided into 4 reaches, which are defined in Table 1 and illustrated in Figure 1. This monitoring program focuses on Reach 2 – 4.

Table 1: Reach break designations and descriptions for the Lower Bridge River

Reach	Boundary (Rkm) Downstream	Upstream	Description
1	0.0	20.0	Fraser River confluence to Camoo Creek
2	20.0	25.5	Camoo Creek to Yalakom River confluence
3	25.5	36.8	Yalakom River confluence to upper extent of groundwater in-flow
4	36.8	40.9	Upper extent of groundwater in-flow to Terzaghi Dam

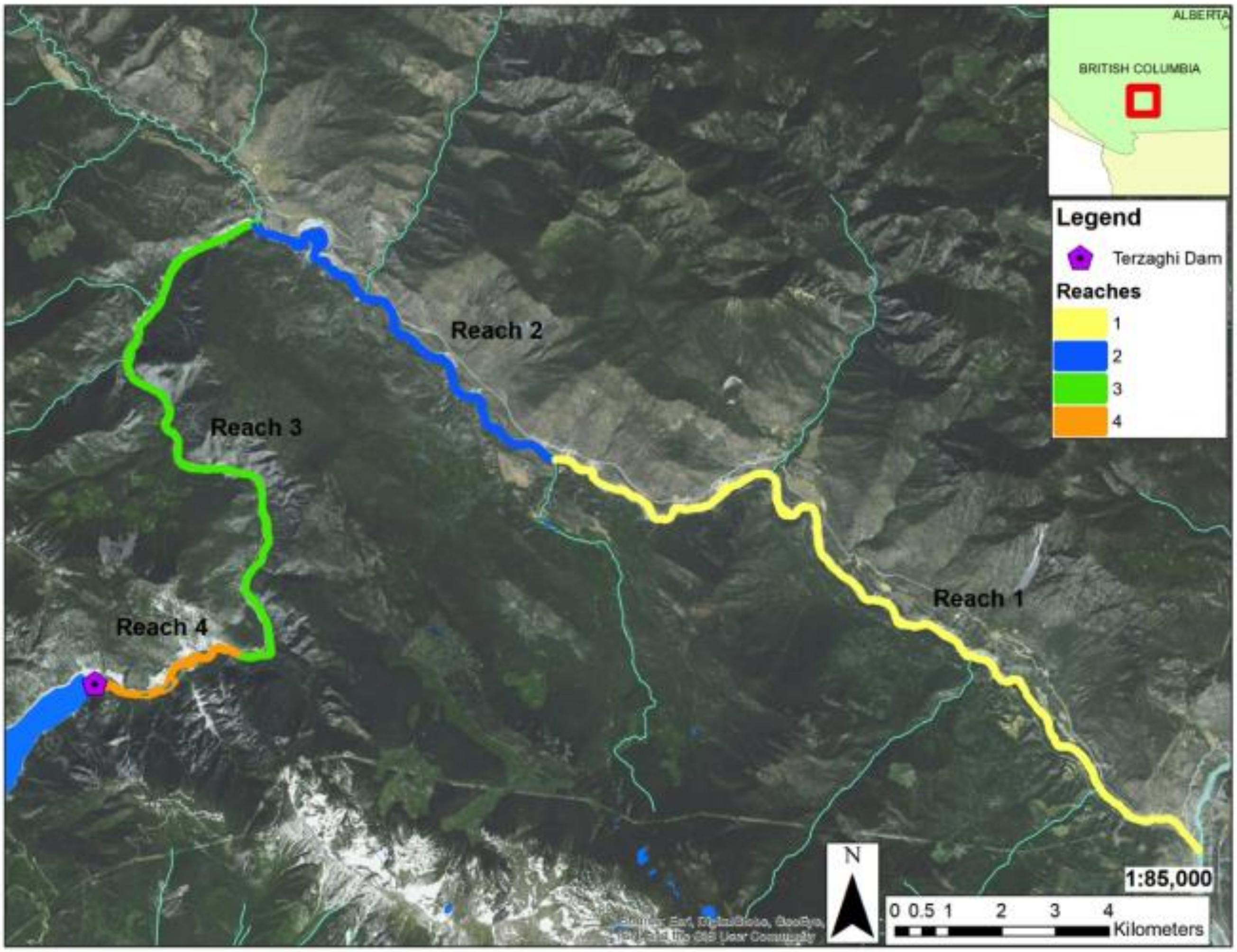


Figure 1: Overview map of the 2016 LBR Aquatic Monitoring Program study area

1.4 Study Period

Aquatic monitoring occurred during nine sampling sessions in 2016. A general description of the activities and sampling timing are presented in

Table 2: Schedule of sampling sessions, 2016

Sample Session	2016 Dates	Activities
Winter	15 February	Water temperature logger downloads
Spring	11 March	Water temperature logger downloads
Summer	15 August; 26 August	Water temperature logger downloads; juvenile growth sampling
Aquatic Habitat Assessment	1 June to 1 August	Digitized spatial habitat
Summer Rampdown	5 to 24 August; 27 and 28 September	Flow rampdown surveys: fish salvage; electrofishing
Fall Stock Assessment	1 to 21 September	Fall standing stock assessment
Early Fall	29 and 30 September	Deploying primary and secondary productivity samplers
Fall	3, October; 7 October; 18 to 21 October	Water chemistry and nutrient sampling; Temperature Logger downloads; Salmonid Juvenile Growth Sampling
Late Fall	28 November; 13 – 16 November; 29 November	Water chemistry and nutrient sampling; salmonid juvenile growth sampling; retrieving primary and secondary productivity samplers
Early Winter	16 December	Temperature Logger downloads

2.0 METHODS

2.1 The Aquatic Monitoring Program

2.1.1 *Overview*

Monitoring methods and protocols utilized in 2016 were standardized to facilitate comparisons across the Trials. These methods and protocols originated from a general template of monitoring initiated at the start of the baseline flow-monitoring phase (1996 – 2000) and have since undergone adaptations through Trials 1 and 2.

- Water temperature,
- River stage,
- Flow release,
- Water nutrient/chemistry,
- Primary productivity (periphyton),
- Secondary productivity (macroinvertebrate),
- Juvenile salmonid growth,

- Fall standing stock,
- Habitat surveys, and
- Rampdown and salvage surveys.

Data collection in 2016 occurred at seven index sites located at 3 km intervals along the LBR, 49 standing stock assessment sites within reaches 2, 3 and 4, and water quality tributary locations (Figure 2). In descending order from Terzaghi Dam, these include the following river kilometers: 39.9, 36.8, 36.5, 33.3, 30.4, 26.4, 26.1, 23.6, and 20.0. The timing and frequency of data collection were similar to historic LBR data collection within the program.

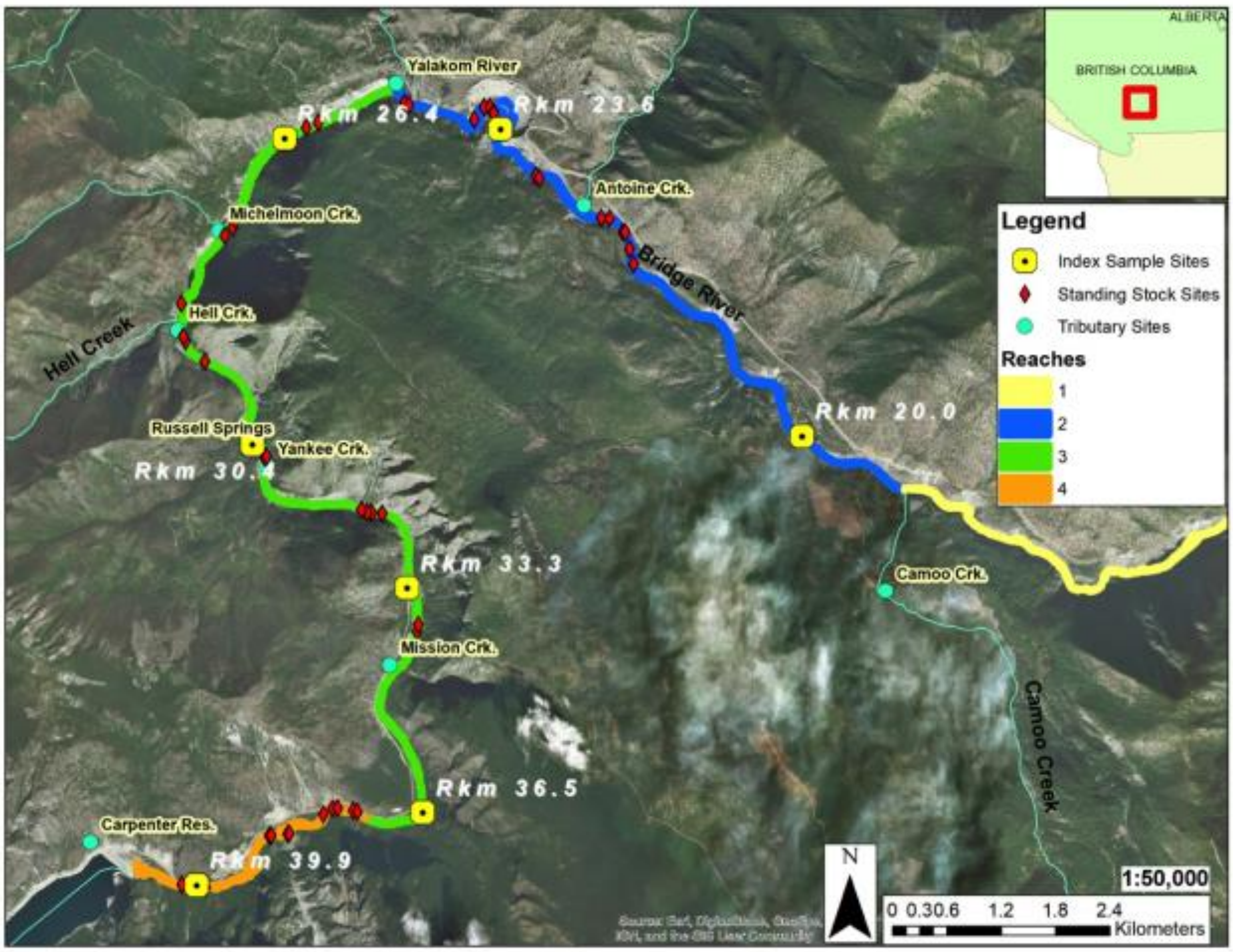


Figure 2: The LBR Aquatic Monitoring Program study area (reaches 2, 3 and 4) index sample site locations, tributaries, and standing stock assessment site locations

2.1.2 *Temperature, Stage and Flow Release*

Water temperature was recorded at an hourly rate on every day of 2016 using UTBI-001 data loggers manufactured by the Onset Computer Corporation (Bourne, MA). These data loggers were located at the seven site index locations as well as an additional logger located in the Yalakom River approximately 100 meters upstream of its confluence with the LBR. Temperature loggers were housed in a protective cover, anchored at locations and submerged to the river bed at a depth of 0 cm. They were both checked and downloaded for data every 3 to 5 months to ensure data quality.

Relative river stage was recorded by PS9000 submersible pressure transducers (Instrumentation Northwest, Inc.), which were coupled with Lakewood 310-UL-16 data recorders. Data were collected at three Rkm locations: 20.0, 26.1, and 36.8. River stage was recorded every 15 minutes per day every day of the year. However, data were irretrievable in Reach 3 and 4 due to ViaSat loggers being displaced from high flows.

Data on flow release were provided by BC Hydro Power Records and are maintained by BC Hydro. These data represent hourly discharge from the Lower Level Outlet (LLO) gates at Terzaghi Dam.

Chinook emergence calculations within the report utilized water temperature data collected using the field methods described above. However, detailed information related to the exact calculations and workflow employed for chinook emergence date predictions can be found in Sneep & Korman (in prep).

2.1.3 *Water Chemistry and Nutrients*

Water chemistry and nutrient data collection occurred in the early fall session on 3 October and 28 November 2016 for the late fall session. During both fall sampling periods, water samples were taken from all seven index locations, Carpenter Reservoir, and the following LBR tributaries: Antoine Creek, Camoo Creek, Hell Creek, Michelmoon Creek, Mission Creek, Russell Springs, Yalakom River, and Yankee Creek (refer to Figure 2). These water samples were submitted to ALS Environmental and analyzed for the following nutrient levels: NH_4 (Ammonium), NO_2/NO_3 (Nitrate/Nitrite), soluble reactive phosphorus (SRP), total dissolved phosphorous (TDP), and total phosphorus (TP); the chemical parameters included total alkalinity and pH. Turbidity (NTU) was also included within in the ALS Environmental analysis. When manual recordings of water were taken they were measured at each site using a WTW handheld field meter (Hanna Instruments, Laval, Quebec) and these included conductivity, pH, and spot water temperature.

2.1.4 *Primary and Secondary Productivity Sampling*

Data were collected in order to assist in the characterization of both spatial (between reaches) and inter-annual variations of primary and secondary productivity. Productivity refers to the rate of generation of biomass in an ecosystem. Primary productivity was monitored using periphyton accrual (chlorophyll-a) as the main indicator parameter. Macroinvertebrate abundance and diversity were the main indicators of secondary productivity. Abundance, when discussed in this report relates to the overall number or count of individuals within a given population (i.e. sampling basket at a specific location in the river). Diversity is defined as the number of taxa (in this case, families) in that population. At each of the seven index site locations, both periphyton and macroinvertebrate data were collected at three replicate subplot locations spaced approximately 20 m apart.

The medium used to accrue periphyton consisted of a 30 x 30 x 1 cm cell Styrofoam sheet that was rubber banded to a plywood backing which was bolted to a 30 x 30 x 10 cm concrete block. At each site index, periphyton accrual samplers were placed at each replicate in areas relatively similar in water depth and velocity. Periphyton accrual data were collected approximately every week at all the replicate subplots and for all seven site index locations between 7 October and 28 November, 2016. Each weekly sample involved the removal of a core of Styrofoam using the open end of a 7-dram plastic vial (8.5 cm^2 core area). These samples were then sent to ALS Environmental for measurement of Chlorophyll-a concentration. Periphyton density and biovolume were not assessed in 2016 due to a sampling error.

The medium used to measure macro-invertebrate abundance and diversity included a standardized metal basket filled with river gravel and substrate collected at each site. These prepared baskets were placed at similar water depths and velocities at each of the site locations and proximal to the periphyton accrual samplers. The baskets were left undisturbed for the duration of the eight-week fall field ecology sampling series at which point they were carefully lifted out of the water and placed into buckets. The contained substrates were carefully removed from the baskets and were hand scrubbed in order to remove all attached material. This material was filtered through a mesh sieve (Nitex), and placed into a sample jar that contained 10% formalin solution. As was done in previous years, the sample jars were sent to Mike Stamford at Stamford Environmental to be sorted, identified to family, and enumerated.

Several benthic invertebrate performance metrics were compared in order to determine differences between the flow trial periods and between the LBR reaches (Stamford 2017). Results were produced utilizing standardized methods and procedural statistical compilation methods outlined in detail in Stamford (2017). These procedural statistics included: mean total abundance, % and number of EPT, (the percentage and total number of families belonging to the mayflies, stoneflies, and caddisflies, EPT taxa), and the Simpsons Diversity Index. The EPT index is generally based on the premise that higher water quality will have a higher % EPT. The Simpsons' Diversity Index is a measure of biodiversity, and incorporates the number of species present and the abundance of each species.

2.1.5 *Sampling for Juvenile Salmonid Growth Data*

In 2016 juvenile salmon were collected for growth data at each index site four times (August, September, October and November) in order to characterize temporal and spatial patterns of fish growth. The intent of this sampling was to collect a target of approximately 30 salmonids within each age/species class. Live fish were collected using backpack electrofishing. Fish were anaesthetized and identified to species. Forklength (mm) and weight (g) measurements were recorded. Following a brief recovery, all fish were released close to their initial collection area.

2.1.6 *Fall Standing Stock Assessment*

The objective of the fall standing stock assessment is to estimate the abundance and distribution of juvenile salmon in reaches 2, 3, and 4. Relative to the fish growth sampling, the standing stock assessment employs a more intensive level of effort, spanning 49 sites along the LBR. The fall stock assessment was conducted during the 3 $m^3 \cdot s^{-1}$ fall flow, a similar season and water flow as sampled in previous years.

Upon arrival to each site, the standing stock survey area was enclosed with three ¼-inch mesh stop nets in size ranging from 50 to 150 m^2. Perpendicular to the bank, two shorter panels were used as stop nets upstream and downstream of the bank while a longer net was used parallel to the bank. Stop nets were attached to bipods and anchored down to the shore so that they were fixed during sampling. As crews changed over the years and the river changed, net placement deviated slightly between crews and depended on site conditions at the time of sampling. This is minimized to ensure that no sampling biases occur.

A four-pass depletion method using LR-24 electrofishers (Smith-Root, Vancouver, WA) was executed within the netted enclosure by using a 400 volts DC. Live fish were collected using backpack electrofishing. Fish were anaesthetized and identified to species, and fork length (mm) and weight (g) measurements were recorded. Fish were kept in a live basket in the stream until the sampling was complete and fish were then released near the original electrofishing location.

Physical (abiotic) data of the site was measured and recorded. Three length and width measurements of the netted enclosure were recorded. The length and width measurements were taken in order to calculate the area sampled. After the net enclosure was removed, water depth and flow velocity was recorded via three transects at upstream, mid, and downstream locations. At each transect five depths and five velocities were measured at equidistant intervals from bank to the offshore extent of the sampled area. Water velocity was measured with an E-230-Model 2100 current meter (Swoffer Instruments, Burnaby BC) at 0.6 of depth. Maximum depth and velocity were also noted at each site. Supplementary site data included sampling effort (electrofishing seconds), date, dominant habitat type, D90, substrate composition, and mean particle size.

Data were compiled and analyzed according to Hierarchical Bayesian Model (HBM) outputs (Korman, 2017; Sneep and Korman, in prep). The HBM was developed for BRGMON-1 to estimate reach-wide standing crop and account for differences in catchability among flow treatments (see Bradford et al. 2011, BCH, 2012):

$$Fij = \mu + \_T + Sj + Yi(T) + eij$$

where, Fij = standing crop biomass in year i at site j; μ is the mean density; _ is the treatment coefficient, T is the fixed treatment effect (dam release), and Yi and Sj are random year and site effects, respectively. Outputs were presented across the years for this report, however significance testing was not conducted due to program and budgetary constraints.

2.1.7 *2016 Aquatic Habitat Assessment Methods*

2.1.7.1 Overview

A single habitat survey was conducted in 2016 and focused on the peak flow of 97 $m^3 \cdot s^{-1}$. Work took place in the period of 15 June through 01 August. The intent of this habitat survey was to complete an office-based and remote sensing spatial habitat classification represented in the form of a geodatabase. The geographic extent of this aquatic habitat survey included reaches 1, 2, 3 and 4. Due to the overall size of the area, the limited time available, and the safety risks associated with the high flows, no field data were collected for validation or to be appended as attribute data (water velocity, depth, width, etc).

2.1.8 *Remote sensing and digitization methods*

The main objective of the 2016 aquatic habitat assessment was to construct a geodatabase that classified aquatic habitat at the 97 $m^3 \cdot s^{-1}$ flow. The principal remote sensing dataset utilized as a reference to this work were orthophotos supplied by BC Hydro. These orthophoto images were captured via airplane in mid-June, 2016.

Initial heads-up digitization of aquatic mesohabitats was employed on the orthophotography in order to map the habitat types. Heads-up digitization is a widely accepted approach to aquatic habitat classification (Northwest Hydraulic Consultants, 2009; Thomson et al., 2001) whereby one uses background imagery (orthophotos) and its characteristics (e.g., a river and its associated habitat types) to trace relevant features. Aquatic features were digitized directly from the aerial photos using ArcMap 10.3.1 (ESRI, 2015). Heads-up digitization of habitat classes was achieved through visual interpretation at an approximate scale of 1:1,000 using a combination of features that included water colour, visible white-water and apparent water flow, substrate, river shape, and riparian vegetation. The remote sensing digitizer used the aforementioned combination of visible aquatic features to determine the proper mesohabitat size, shape and classification which would take the form of a habitat unit or subunit. A habitat unit consisted of a mesohabitat that was characterised by similar aquatic characteristics. A habitat subunit was defined as small areas of habitat within the larger habitat unit but with distinct physical characteristics. These habitat subunits were classified as part of the main habitat unit but were given their own unique identifier. Different habitat unit and subunit classes with their descriptions are outlined in Table 3. Aquatic mesohabitat class types were taken from historical field methods utilized in the LBR (Sneep, 2012) as a means for data consistency and comparability across annual LBR habitat surveys. The final spatial product includes a geodatabase representing the LBR classified by aquatic habitat types with an emphasis on habitat types important to salmonid species and the size and shape of each habitat unit applicable subunit.

Table 3: Outline of descriptions and definitions utilized to identify habitat types

Habitat Type	Depth	Velocity	Gradient	Instream Cover	Comments
Run	Mod. to High	Mod.	Low to Mod.	Mod.	Moderate, laminar flow; little surface agitation
Riffle	Low to Mod.	High	Mod. to High	Mod. to High	Swift, turbulent flow; some partially exposed substrate
Pool	High	Low	Low	Low to High	Variety of forms; can be either 1º or 2º units
Cascade	Mod	High	High	Low	Very steep riffle habitat; Substrate is usually boulders
Rapid	Mod. to High	High	Mod.	Low	Very fast flowing runs, flooded riffles; Around constrictions
Side-channel	Low to Mod.	Low to Mod.	Low to Mod.	High	2º habitat type; productive but limited quantity in LBR
Bar	N/A	N/A	N/A	N/A	Can be wetted or dry and usually lacks vegetation
Island	N/A	N/A	N/A	N/A	Always dry and normally have annual and perennial vegetation.

2.1.9 *Flow Rampdown Surveys*

Flow rampdown and stranding risk surveys were limited to <15-$m^3 \cdot s^{-1}$, and data summarizing rampdown results for flows > 15 $m^3 \cdot s^{-1}$ can be found in the LBR Aquatic Monitoring (2016 High Flow) report (McHugh et al, 2017).The focus study area of the LBR rampdown occurred between Terzaghi Dam and the confluence of the Yalakom River, a river length of 16 km. At the start of each rampdown day, a preliminary baseline reconnaissance of the entire 16 km was conducted. The physical progress of the flow reduction was monitored according to the BCH

LBR Fish Stranding Protocol (Sneep, 2016) and close attention was paid to those areas with historically high fish stranding potential.

Based on historical data, reporting, and stage levels for the rampdown component, potential areas with risk were identified daily, and salvage crews were dispatched to those areas. Upon arrival, these crews documented the physical attribute characteristics of the area; and if necessary, crews began fish salvage. At the start of the work day, fish salvage efforts started closest to Terzaghi Dam and highest priority was given to the following river habitats: sidechannels, low gradient edge habitats, and 'potholes' from historical gold mining endeavours.

2.1.10 *Fish Salvage*

When crews arrived to an identified fish salvage site, physical habitat attribute information was recorded as noted. These notes included:

- Date, time, full name of crew members, operational changes being assessed,
- General site description (i.e. reach #, river km, bank location, proximity to landmarks, etc.),
- NAD 1983 UTM Zone 10 North coordinates,
- Estimated dewatering time for the site, and
- Additional comments.

Upon arrival at each site, crews assessed the area for presence of fish, and estimated the size of habitat that would likely dewater. A strategy for moving fish out of the affected area and back into the main river was determined. During active salvage, fish were categorized according to the following categories:

- Incidental - fish habitats that were not yet isolated, and fish still had the opportunity to move to deeper areas on their own;
- Isolated – fish in wetted areas that were isolated from the main flow of the river (i.e. strand pools)
- Stranded – fish that were found in habitats that had completely dewatered, but were still alive when salvaged;
- Mortality – fish that were found dead in habitats that were isolated or completely dewatered.

Fish that were salvaged from shallow waters within potential stranding areas prior to complete isolation from the main channel were considered 'incidental' captures. Crews counted and recorded the total number of incidental captures within this category. When sites were completely isolated from the main channel and fish could not be captured in an incidental manner, they were captured by backpack electrofisher. All captured fish were categorized, counted and identified to species before returning them back to the main channel. A subset of the captured fish were measured for forklength (to the nearest mm).

At the end of each ramp-down event, an update was provided to BC Hydro regarding the environmental monitoring conducted on the LBR, including: rampdown stages, flow reduction, fish salvage crew numbers, the number and location of sites used for salvage and the total number of fish salvaged in relation to salvage category.

3.0 AQUATIC MONITORING RESULTS

3.1 Physical Conditions

3.1.1 *River Stage*

The mean annual discharge from Terzaghi Dam (TRZ) in 2016 was ~22 $m^3 \cdot s^{-1}$ (21.86 $m^3 \cdot s^{-1}$). The hydrograph ranged from a spring and summer peak of approximately 97 $m^3 \cdot s^{-1}$ during June to a fall and winter low of approximately 1.5 $m^3 \cdot s^{-1}$ (Figure 3).

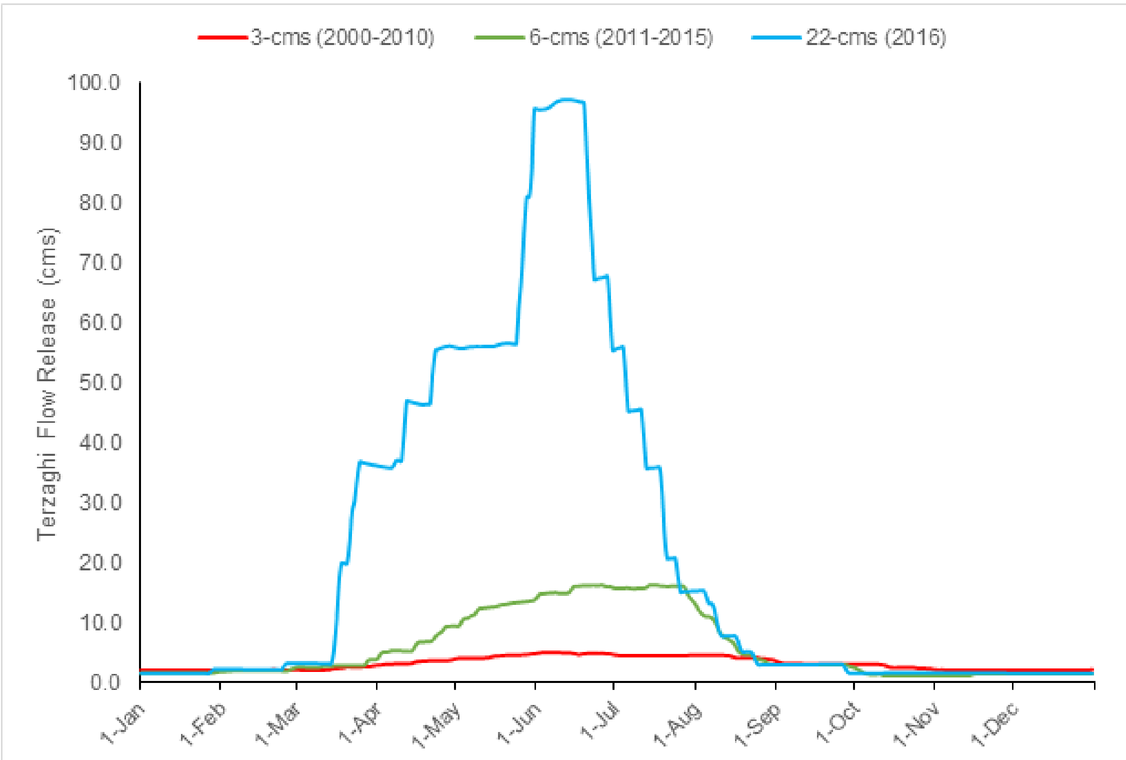


Figure 3: Lower Bridge River hydrographs during the 3 $m^3 \cdot s^{-1}$ Trial 1 period (2000-2010), the 6 $m^3 \cdot s^{-1}$ Trial 2 period (2011 – 2015), and 22 $m^3 \cdot s^{-1}$ in 2016

Staged ramp-up from 3 $m^3 \cdot s^{-1}$ to 15 $m^3 \cdot s^{-1}$ began on 26 February and progressed within WUP target values. Starting on 15 March, flows diverted from the WUP Trial 2 hydrograph and peaked at a discharge of 97 $m^3 \cdot s^{-1}$. Between 20 June and 26 July, eight rampdown events took place to restore flow to WUP targets. Flows were restored to 15 $m^3 \cdot s^{-1}$ on approximately July 25 (Figure 3). Between 5 August and August 24, flow release was ramped down in stages from 15-$m^3 \cdot s^{-1}$ to 3 $m^3 \cdot s^{-1}$. These stages occurred across multiple weeks due to a flow change of approximately 12 $m^3 s^{-1}$. Between 27 and 28 September the LBR was further ramped down from 3 $m^3 \cdot s^{-1}$ to 1.5 $m^3 \cdot s^{-1}$.

3.1.2 *Water Temperature*

3.1.2.1 2016 Water Temperature Results

Annual mean daily water temperatures during 2016 for reaches 2, 3, 4 and the Yalakom River are presented in Figure 4.

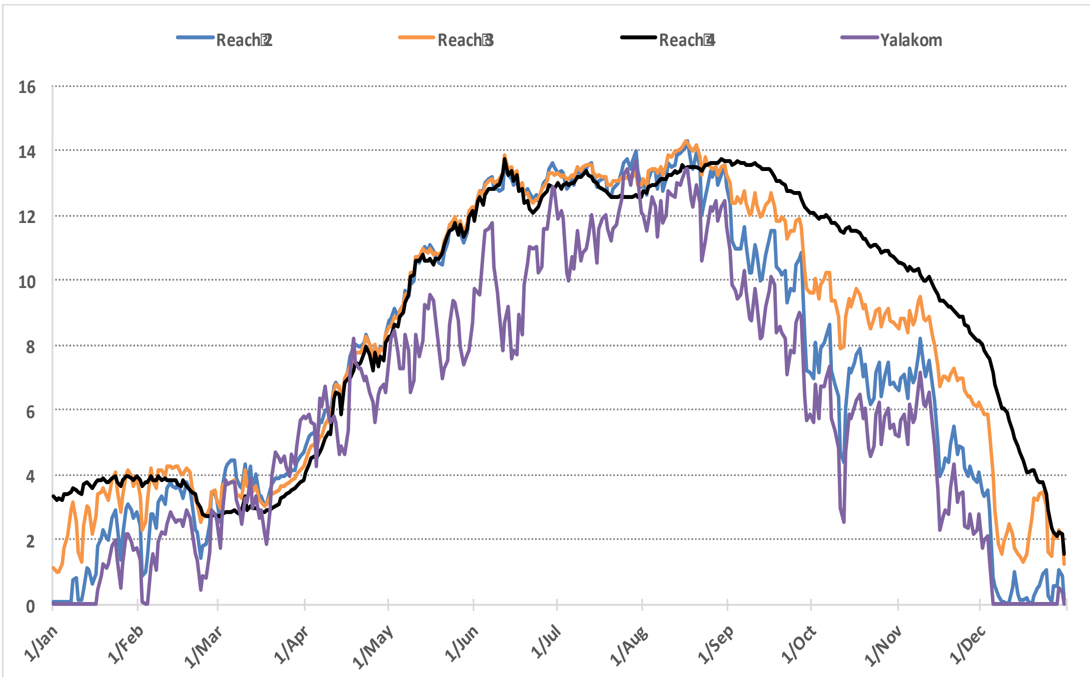
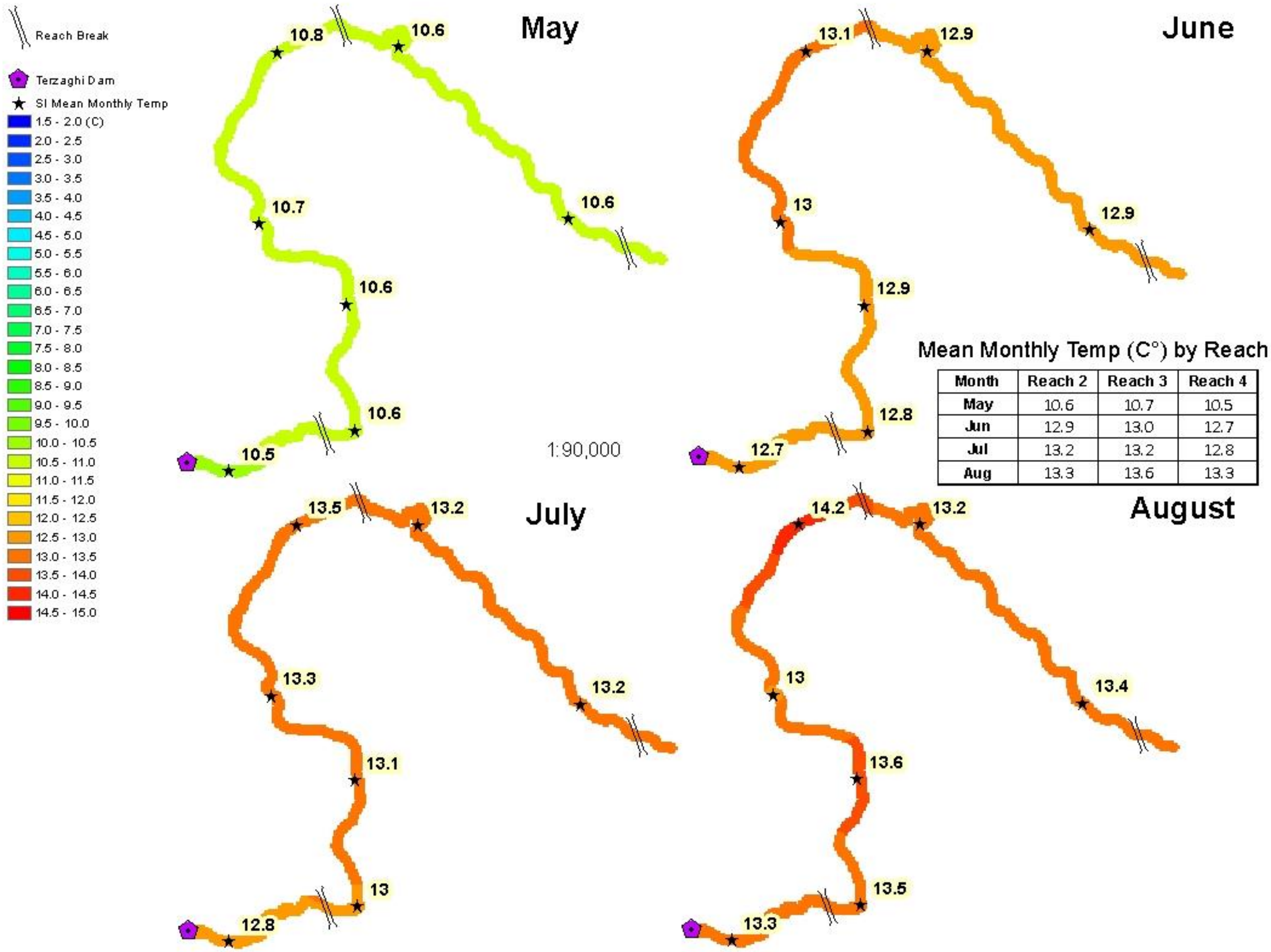


Figure 4: LBR reaches 2, 3, 4 and Yalakom River mean daily temperatures between 1 January and 31 December 2016

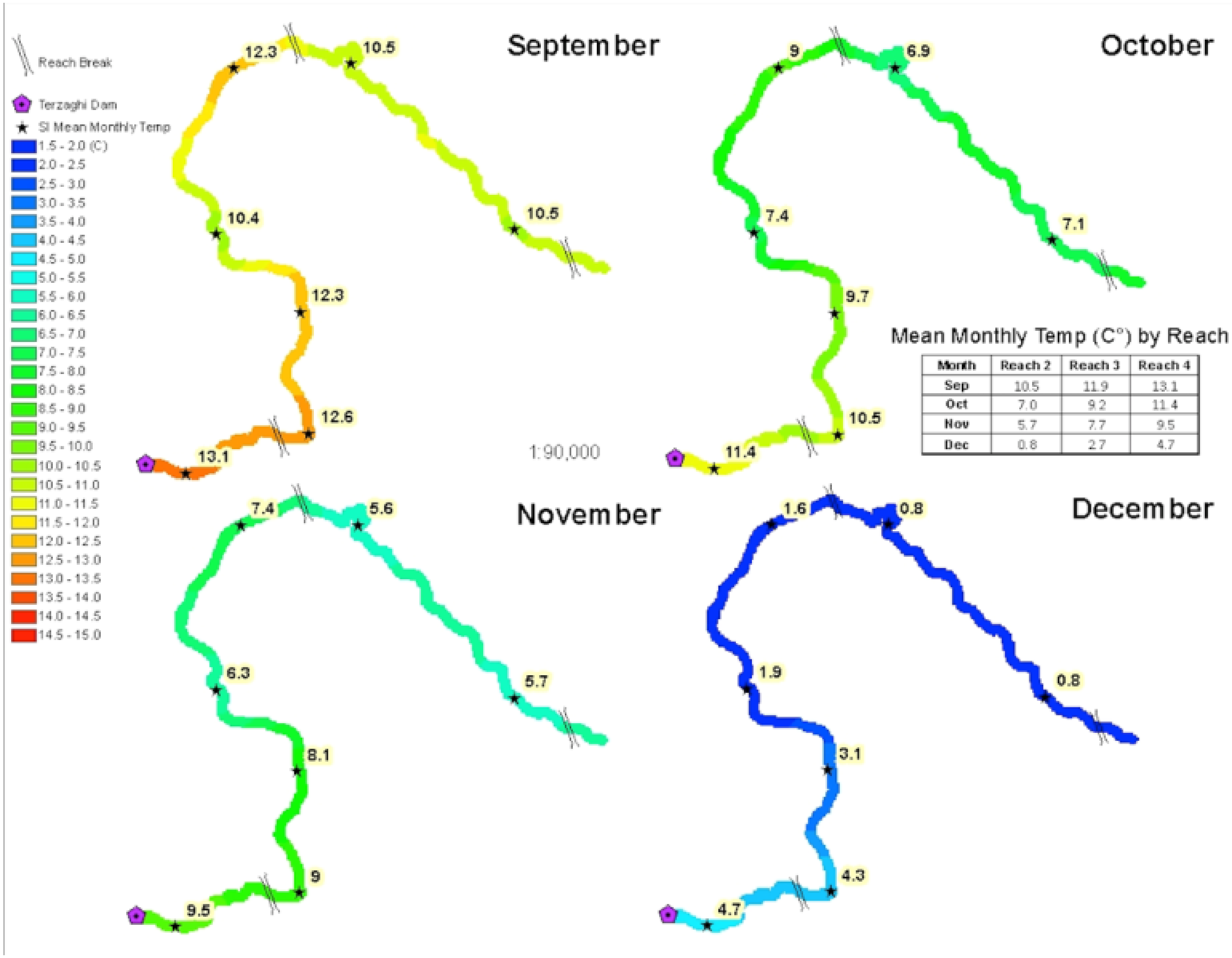
Seasonal temperature trends for 2016 in reaches 2 – 4 of the Lower Bridge River were similar to those observed throughout Trials 1 and 2 (McHugh & Soverel, 2015; 2014; 2013). The thermal effects of the release exhibit an upstream to downstream gradient that varies by season. Figure 5 and Figure 6 geographically display a colour ramp indicating mean monthly temperature, by reach, during the winter, spring and summer seasons.

The contributing factors that set up the thermal profile of the LBR are the temperature of the reservoir, volume of the release, time of exposure to ambient influence (distance from the dam), and attenuation of tributary inflows. Spring and summer temperatures in 2016 were higher than average observed through both Trials 1 and 2 (Figure 7). In general, temperatures in reaches 3 and 4 were warmer in the fall and early winter compared to the Pre-Flow thermal regime (Figure 7). Across the fall period, temperatures in Reach 4 were on average 3.6°C warmer than Reach 2 (Figure 4). Temperature reflected the principal influence of the hypolimnetic flow from the reservoir through reaches 2, 3 and 4, with effects that extended farther downstream during the June and July high flow periods (Figure 5). Outside of the high flow period, temperatures in Reach 2 were moderated by the influence of the unregulated Yalakom River and other tributaries, groundwater influences (Figure 4 and Figure 7; Appendix A., Figure 1) and differing channel morphology such as steep shaded canyon walls and a steeper gradient.



Month	Reach 2	Reach 3	Reach 4
May	10.6	10.7	10.5
Jun	12.9	13.0	12.7
Jul	13.2	13.2	12.8
Aug	13.3	13.6	13.3

Figure 5: Temperature schematic of mean monthly water temperature (C°) recorded at each site index location along the LBR in May, June, July and August of 2016. Site indices on the map are in order from upstream to downstream (Rkm): 39.9, 36.5, 33.3, 30.4, 26.4, 23.6 and 20.0 The colour ramp represents warmest water temperatures with shades of red and decreasing water temperatures progressing into orange and yellow, followed by green and finally the dark blue colour represents the coldest temperatures



Month	Reach 2	Reach 3	Reach 4
Sep	10.5	11.9	13.1
Oct	7.0	9.2	11.4
Nov	5.7	7.7	9.5
Dec	0.8	2.7	4.7

Figure 6: Temperature schematic of mean monthly water temperature (C°) recorded at each site index location along the LBR in September, October, November and December of 2016. Site indices on the map are in order from upstream to downstream (Rkm): 39.9, 36.5, 33.3, 30.4, 26.4, 23.6 and 20.0. The colour ramp represents warmest water temperatures with shades of red and decreasing water temperatures progressing into orange and yellow, followed by green and finally the dark blue colour represents the coldest temperatures

3.1.2.2 Water Temperature: Trial Comparison

Annual mean daily temperature trends during Pre-Flow (1996-2000) Trial 1 (2000-2010), Trial 2 (2011-2015) and 2016 are presented in Figure 7. In 2016, Trial 1 and Trial 2, spring and fall temperatures were distinctly warmer, and summer temperatures were consistently cooler than observed in the Pre-Flow period. Like 2016, across the trials, these effects were strongest in the upper reaches (reaches 3 and 4) and weakest in Reach 2, reflecting the primary influence of the hypolimnetic flow from TRZ. In 2016, warmer water temperatures remained above the Pre-Flow, Trials 1 and 2 between May and August (Figure 7).

The thermal regime produced increased fall temperatures of approximately 2 – 4 C° relative to the Pre-Flow baseline, particularly in reaches 3 and 4 (Figure 4, Figure 5, Figure 6). This time period coincided with the annual LBR chinook egg incubation period. Implications of the altered thermal regime on the emergence and subsequent survival of chinook fry are discussed in Section 3.3.2.4. Coho and rainbow trout were likely not impacted by the elevated temperatures, as egg development occurs during a different season. Throughout this report, juvenile O. mykiss are referred to as rainbow trout, although a large and undefined proportion of these fish in the LBR are anadromous steelhead. Temperatures changes in other seasons were minimal, and were not observed to impact juvenile salmon life cycles.

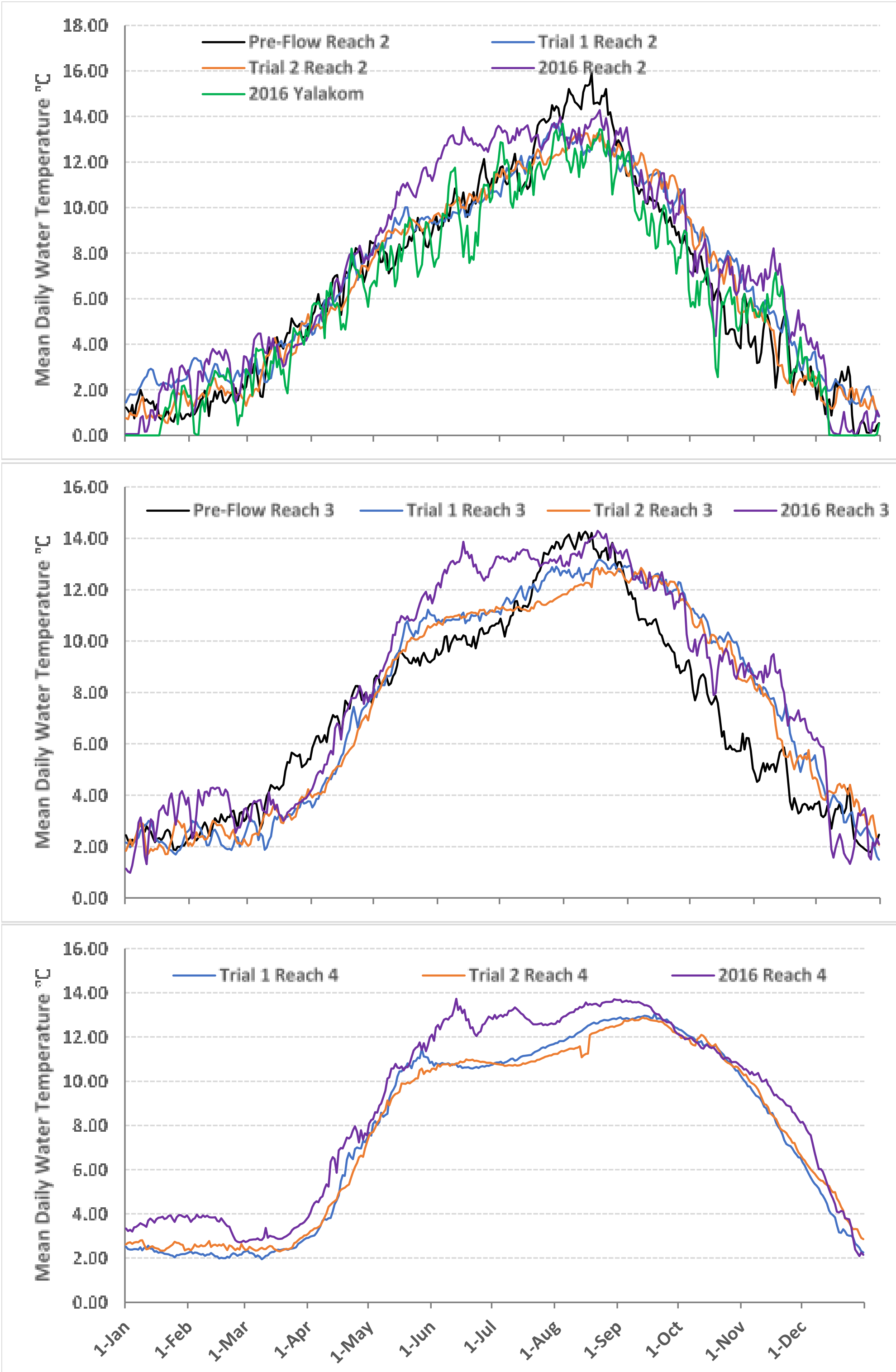


Figure 7: Pre-Flow, Trial 1, Trial 2, and 2016 comparisons for LBR reaches 2 (top), 3 (middle) and 4 (bottom) figures. No Pre-Flow data are presented in Reach 4 because there was no water during that time period (1996 – 2000)

3.1.3 *Water Chemistry*

Water chemistry samples were collected from the LBR, Carpenter Reservoir, and tributaries within the study area during October 3, 2016 and November 28, 2016. The water chemistry parameters observed in 2016 were similar to those reported in previous non-pink salmon spawning years, and differences were minimal. All levels of parameters measured were within the normal range and within British Columbia Approved Water Quality Guidelines: Aquatic Life, Wildlife and Agriculture (Ministry of Environment 2017). The Lower Bridge River is an alkaline environment. The levels of pH in the main stem remained in the optimal category for most organisms and ranged from 7.72 to 8.19. Tributaries levels ranged from 8.19 - 8.23, and Carpenter Reservoir pH remained consistent at 7.7.

Turbidity levels in the LBR ranged from 1.41 to 20.4 NTUs, with Carpenter Reservoir measuring 4.40 (early fall) and 6.15 NTUs (late fall). Turbidity levels in the tributaries ranged from 0.1 to 1.17. Concentrations of nitrates and phosphate levels were within the British Columbia Approved Water Quality Guidelines: Aquatic Life, Wildlife and Agriculture, and remained relatively stable through 2016, and since the Flow Trials began. In early fall sampling, results showed a decline or stabilization in all parameters, with the exception or a rise in turbidity, within all reaches of the main stem compared to Pre-Flow, Trials 1 and 2. In late fall sampling there was also a general decline and stabilization of all parameters comparable to Pre-Flow, Trial 1 and Trial 2. As such, these differences cannot be easily distinguished from natural variations between years using descriptive graphical comparison. A more rigorous statistical comparison should be conducted in future years to determine if water quality should continue to monitored twice annually during the fall field ecology series, or if monitoring during other seasons, like high summer freshet, would be more beneficial for reducing uncertainties in the management questions.

3.1.4 *Aquatic Habitat Assessment Results*

An increase in wetted area occurred at the 97 $m^3 \cdot s^{-1}$ when compared to all other flow regimes (Table 4). Total wetted area across reaches 2, 3, and 4 at 97 $m^3 \cdot s^{-1}$ was ~830,000 m^2 compared to 541,000 m^2 at 15 $m^3 \cdot s^{-1}$. The largest increase in wetted area per habitat type between the 15 and 97 $m^3 \cdot s^{-1}$ occurred in cascade/rapid habitat, from ~75,200 m^2 at the 15 $m^3 \cdot s^{-1}$ to 292,500 m^2 at the 97 $m^3 \cdot s^{-1}$ (Table 4, Figure 8). Sidechannel habitat area increased from 17,700 m^2 to 35,500 m^2 (Table 4). Riffle, run and pool habitat also increased, but these increases were relatively minor when compared to the increase in total wetted area. Relative proportions of cascade/rapid habitat increased by greater than 20% (from 14% at the 15 $m^3 \cdot s^{-1}$ to 35% at the 97 $m^3 \cdot s^1$); decreased for riffle habitat (from 52% at the 15 m^3/s to 38% at the 97 $m^3 \cdot s^{-1}$) and run habitat (from 27% at the 15 $m^3 \cdot s^{-1}$ to 18% at the 97 $m^3 \cdot s^{-1}$) by greater than 10%, and were similar for sidechannel (from 3% at the 15 $m^3 \cdot s^{-1}$ to 4% at the 97 $m^3 \cdot s^{-1}$) and pool habitat (from 4% at the 15 $m^3 \cdot s^{-1}$ to 4% at the 97 $m^3 \cdot s^{-1}$) (Table 4, Figure 8). At the other end of the flow magnitude spectrum, proportion of habitat type and total area were similar across the lowest flows (1.5 $m^3 \cdot s^{-1}$, 3 $m^3 \cdot s^{-1}$) suggesting they contained a similar amount of rearing habitat for juvenile salmon.

Table 4: Aquatic habitat survey results, depicted as total area, 100 m^2, conducted between 1996 and 2016 for various flows in the LBR. A dash indicates data were unavailable

Reach	Habitat Type	Sep-96 0 m3·s-1	Jul-00 0 m3·s-1	Oct-13 1.5 m3·s-1	Oct-14 1.5 m3·s-1	Oct-06 3 m3·s-1	Sep-15 3 m3·s-1	Aug-00 4 m3·s-1	Jun-07 5 m3·s-1	Jul-07 8 m3·s-1	Jul-14 15 m3·s-1	Jun-16 97 m3·s-1
4	Run	-	-	140	195	149	168	145	83	141	146	215
	Riffle	-	-	247	286	310	227	489	363	346	230	297
	Pool	-	-	190	186	223	205	120	222	260	196	224
	Ca/Rapid	-	-	-	2	-	39	-	55	61	213	431
	SC	-	-	41	29	37	41	37	55	72	35	92
	Subtotal	**-**	**-**	**618**	**697**	**718**	**680**	**792**	**778**	**880**	**821**	**1,260**
3	Run	618	581	630	798	543	784	818	730	838	771	693
	Riffle	1004	1211	1296	1278	1569	1236	1186	1449	1297	1288	1712
	Pool	52	54	176	114	183	147	71	174	124	3	69
	Ca/Rapid	89	93	-	11	23	11	30	442	482	344	1379
	SC	-	-	39	70	2	50	2	45	48	109	153
	Subtotal	**1,763**	**1,939**	**2,141**	**2,272**	**2,319**	**2,229**	**2,107**	**2,839**	**2,789**	**2,514**	**4,006**
2	Run	541	208	-	752	605	-	555	580	-	557	586
	Riffle	1093	1581	-	975	917	-	1288	591	-	1282	1163
	Pool	18	18	-	8	12	-	6	15	-	13	60
	Ca/Rapid	87	105	-	95	254	-	76	901	-	195	1116
	SC	71	71	-	94	87	-	87	124	-	33	110
	Subtotal	**1,809**	**1,983**	**-**	**1,924**	**1,876**	**-**	**2,013**	**2,211**	**-**	**2,079**	**3,034**
1	Run	-	-	-	-	-	-	-	-	-	-	490
	Riffle	-	-	-	-	-	-	-	-	-	-	2459
	Pool	-	-	-	-	-	-	-	-	-	-	113
	Ca/Rapid	-	-	-	-	-	-	-	-	-	-	2348
	SC	-	-	-	-	-	-	-	-	-	-	139
	Subtotal	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**5,549**

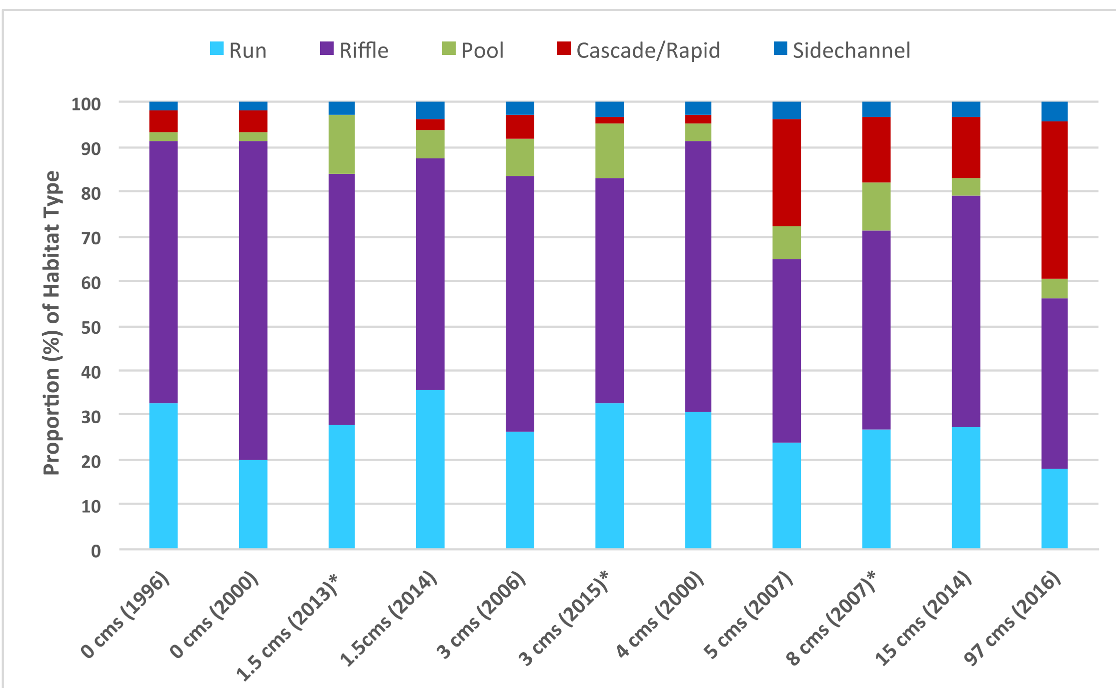


Figure 8: Proportion of habitat types within the LBR in reaches 2, 3 and 4 for each studied flow discharge, by year and habitat survey. An * indicates that no data were available for Reach 2 during that survey year

Higher flows provided more wetted area, but much of the additional habitat also likely contained increased velocities relative to the 15 m^3/s and other WUP flows, and it is unclear if fish were able to access and utilize newly wetted rearing habitat areas. Visualization of the spatial habitat results indicated that cascade/rapid habitat type became more numerous and prolific throughout reaches 2, 3 and 4. Both the proportion of cascade habitat and the distribution of cascade/rapid habitat increased throughout the LBR (Figure 8; Figure 9). This may have caused displacement of fish, or impacted migration and movement patterns of juveniles. Based on predictions from habitat suitability curves produced for the LBR (Sneep and Korman, In prep), results in 2016 indicated that the increase in wetted area corresponded with a reduction in the suitability of instream habitats, particularly in the cascade/ rapid habitat type, as flow velocities under higher flows were increased to above optimal thresholds for rearing juvenile salmon through large portions of the river.

These results were further supported by an additional research component that focused on the identification of potential enhancement sites to act as refugia habitat for rearing salmon at high flows within the LBR in 2017 (McHugh et al., 2017). Few locations within the river were predicted to have high rearing habitat suitability or enhancement potential under the 97 m^3s^{-1} flow. Overall, habitat assessments conducted across LBR monitoring programs in 2016 and 2017 suggested a reduction in the quality and quantity of available rearing habitat under high flows.

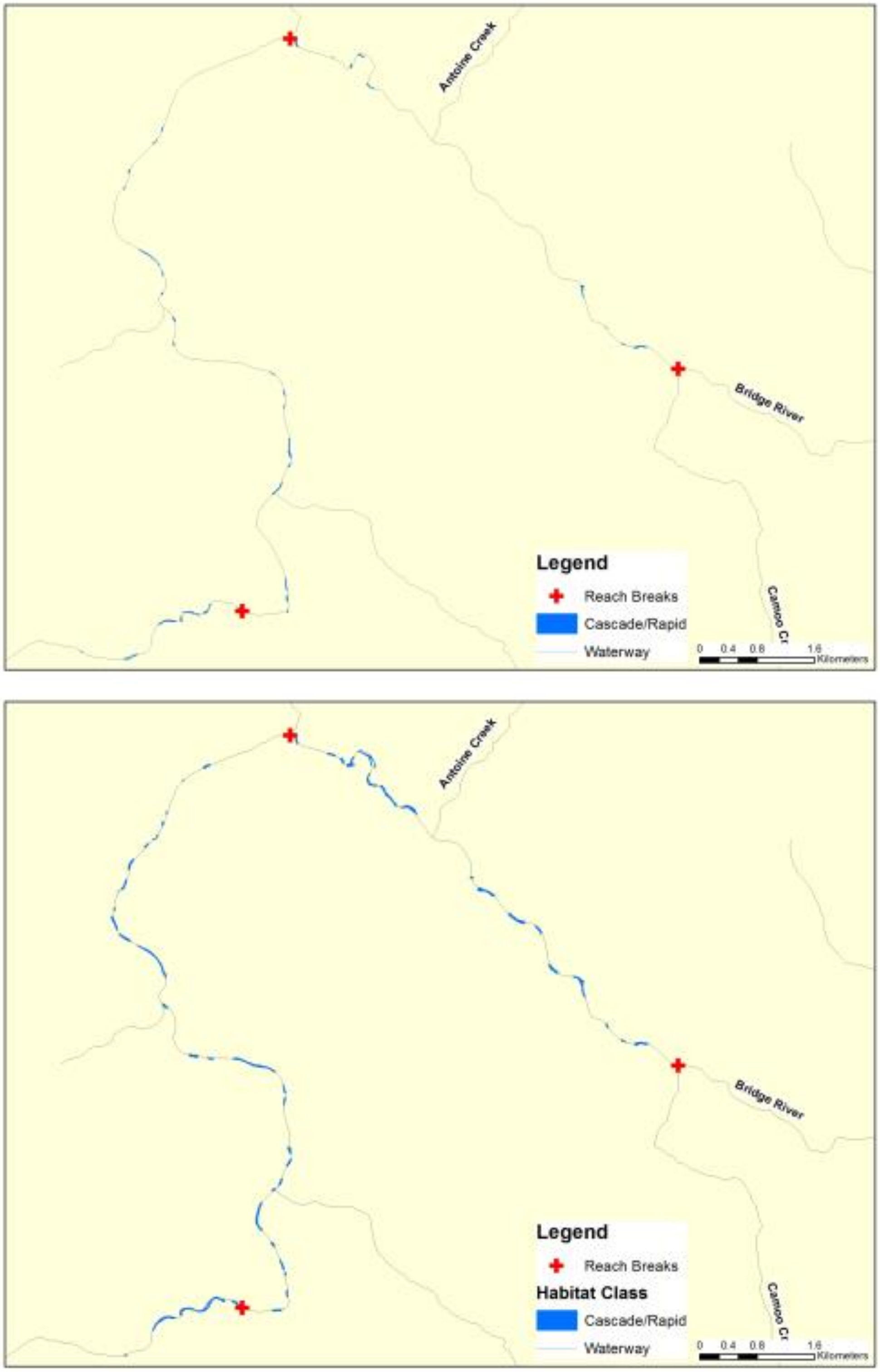


Figure 9: Distribution of cascade / rapid habitat type within reaches 2, 3 and 4 and depicted at the 15 $m^3 \cdot s^{-1}$ flow (top) and 97 $m^3 \cdot s^{-1}$ flow (bottom)

3.2 Periphyton and Macroinvertebrates

3.2.1 *2016 and Trial comparison of Periphyton Results*

Mean periphyton accrual rates (measured as cumulative concentration of chlorophyll-a) for the entire fall sampling period in Trials 1, 2 and 2016 are depicted in Figure 10. Data depict non-pink salmon years (even years) as historical trends of periphyton accrual. Trends, which were demonstrated in even and odd years in previous LBR reports, relate strongly with spawning fish deposition and subsequent accumulation of nutrients (McHugh & Soverel, 2015; Sneep & Hall, 2012).

Results indicated that in 2016 reaches 3 and 4 showed relatively similar accrual patterns for a non-pink year when comparing to Trials 1 and 2 (Figure 10). This trend typically showed reaches 3 and 4 increased through sample week 8 with Reach 4 accruing more periphyton than Reach 3 and Reach 2 (Figure 10). Reach 3's 2016 results show a lower overall accrual when compared to Trials 1 and 2 (Figure 10). Reach 2's trend for non-pink years is very gradual accrual and consistently lower than reaches 3 and 4 throughout the samples weeks. The year 2016 appears to follow a very similar trend when compared to Trials 1 and 2 (Figure 10).

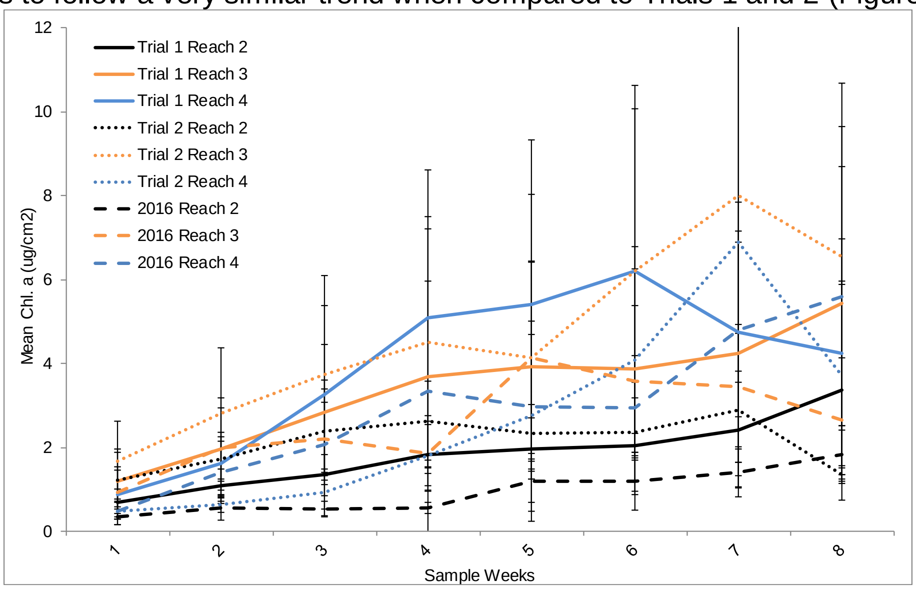


Figure 10: Mean periphyton accrual (measured as chlorophyll-a) on artificial substrates in the LBR, during the fall series sampling in even years in Trials 1 and 2 and the year 2016. Each point represents an average accrual for all stations within a reach; error bars represent (+/-) standard deviation. Samples weeks (1-8) represent weeks between early October and late November in ascending order

3.2.2 *2016 and Trial Comparison of Macroinvertebrate Results*

Macroinvertebrate abundance and biodiversity were the primary metrics used to measure benthic invertebrate health and production within the LBR over the last 20 years. These metrics were compared across differing time periods, including the year 2016 (22 $m^3 \cdot s^{-1}$), Pre-Flow (0-$m^3 \cdot s^{-1}$), Trial 1 (3 $m^3 \cdot s^{-1}$) and Trial 2 (6 $m^3 \cdot s^{-1}$) and between the reaches by Stamford (2017).

Total macroinvertebrate abundance and diversity initially increased following the flow release in 2000 as the benthic invertebrate community across reaches 2 and 3 adjusted to the flow release or became established in reach 4 (Stamford, 2017). Once the ecosystem stabilized, numbers of invertebrates were similar under Trial 1 and Trial 2 (Figure 11). In 2016, following high flows, macroinvertebrate abundance decreased in all reaches and across the five most abundant invertebrate taxa present when compared to total abundance estimates observed during the years in Trial 1, Trial 2 and Pre-Flow periods (Table 5, Figure 11; Appendix A., Table 1, Stamford, 2017). Mean total abundance dropped by approximately 79% in Reach 2, compared estimates during the years in Trial 2, and by 83% (Table 5, Figure 11) compared to average estimates during Trial 1. No differences were apparent in abundance in Reach 2 between 2016 and years in the Pre-Flow period (Table 5, Figure 11). Abundance estimates in Reach 3 dropped in 2016 compared to the trials and most notably by 79% compared to the mean observed during Pre-Flow years (Table 5; Figure 11). Mean total abundance estimates in Reach 4 also declined in 2016 when compared to mean estimates within Trial 1 and Trial 2 (70% and 77%, respectively; Table 5, Figure 11).

While abundance declined substantially in 2016, diversity remained high in the LBR invertebrate community. The total number of EPT families and total number of families increased in Trial 2 (Table 5, Figure 12). EPT % did not appear to change in Reaches 3 and 4 across the trials and through 2016 (Figure 12), with the exception of the increase that occurred following the initial flow release at the start of Trial 1. Higher relative abundances of mayflies (e.g. Heptageniidae, Ephemerellidae) may have lowered the Simpsons index (increased evenness) and increased %EPT in Reach 3 (Table 5, Figure 13), although data were highly variable. Reaches 3 and 4 appear to have similar 2016 biodiversity values (both EPT percent/number and Simpsons Index) when comparing to all previous Trials (Table 5, Figure 12, Figure 13). Relative abundances of chironomids and simuliids (both Diptera) declined proportionally more than other taxa in Reach 2, which resulted in a slightly more even distribution of abundance among taxa (Stamford, 2017). An overall decline in abundance with minimal changes to diversity may suggest the complexity of aquatic habitat did not decline in 2016, and possibly even increased in Reach 2. Continued monitoring at high flows would help to reduce this uncertainty.

Table 5: Mean total abundance and diversity indices among LBR aquatic invertebrate taxa (family level and higher) that colonized basket samples within reaches and among flow periods. The ranges from 95% confidence intervals are shown in parentheses. Table from Stamford, 2017

Reach	Trial	Mean Total Abundance (range, 95% CI)	mean %EPT (range, 95 CI)	mean #EPT Families (range, 95%CI)	mean Simpsons Index (range, 95%CI)
2	0 CMS	922 (181-1436)	66 (55-78)	7.9 (6.7-9.2)	0.28 (0.23-0.32)
	3 CMS	5277 (3943-6402)	41 (36-47)	12.7 (12.0-13.4)	0.31 (0.27-0.34)
	6 CMS	4357 (3028-5520)	46 (41-52)	13.2 (12.5-13.9)	0.29 (0.24-0.32)
	22 CMS	916 (399-1344)	76 (66-87)	12.3 (11.1-13.3)	0.16 (0.15-0.18)
3	0 CMS	4842 (2381-6832)	41 (33-49)	9.1 (8.6-9.6)	0.45 (0.38-0.51)
	3 CMS	3237 (2659-3762)	60 (56-64)	10.1 (9.7-10.6)	0.32 (0.30-0.35)
	6 CMS	3049 (2356-3680)	56 (49-63)	10.9 (10.3-11.6)	0.31 (0.28-0.33)
	22 CMS	1008 (357-1461)	63 (50-78)	9.0 (8.3-9.7)	0.34 (0.27-0.41)
4	0 CMS	0 (0)	0 (0)	0 (0)	0 (0)
	3 CMS	2968 (2348-3528)	52 (45-58)	8.8 (8.2-9.4)	0.41 (0.35-0.46)
	6 CMS	3910 (2283-5262)	47 (34-60)	8.1 (7.1-9.5)	0.41 (0.31-0.50)
	22 CMS	901 (84-1582)	43 (31-57)	7.7 (6.3-10.3)	0.42 (0.30-0.49)

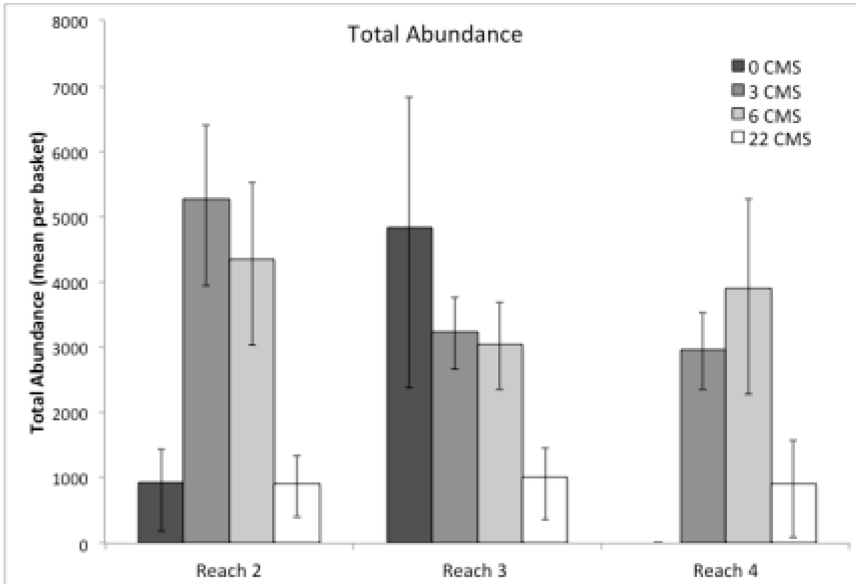


Figure 11: Total Abundance (all taxa combined) of benthic invertebrates in fall baskets within each reach and between flow trials: Pre-Flow, Trial 1, Trial 2 and 2016 (95% confidence intervals determined from simple bootstrap procedure). Figure from Stamford, 2017

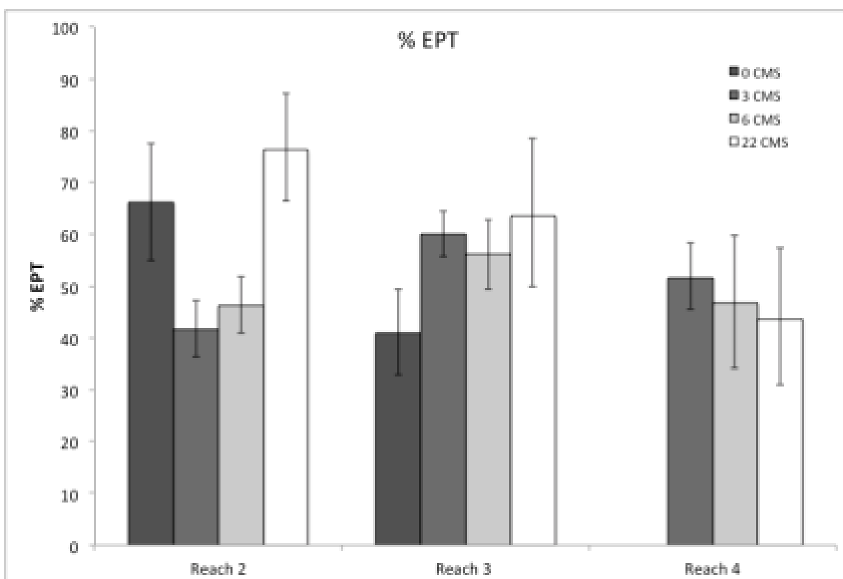


Figure 12: Percent abundance of Ephemeroptera, Plecoptera, and Trichoptera orders combined, relative to total invertebrate abundance (% EPT) among LBR basket samples grouped by reach and flow trial period. Error bars are 95% confidence intervals determined from a simple bootstrap procedure. Figure from Stamford, 2017

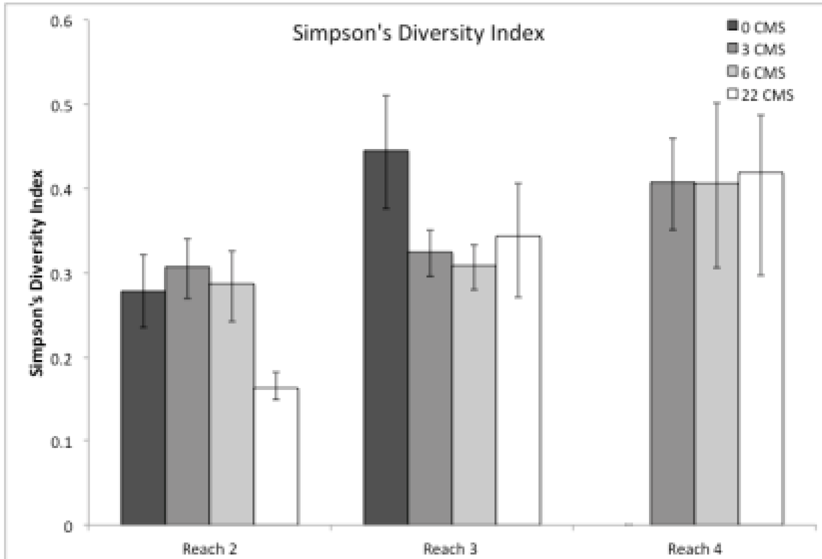


Figure 13: Simpsons diversity index (with 95% bootstrap CI) among fall basket samples. Figure from Stamford, 2017

3.3 Fish Sampling for Abundance and Growth Assessments

A total of 821 fish were sampled during backpack electrofishing during the annual fall standing stock assessment (Reach 4, *n*=151; Reach 3, *n*=335; and Reach 2, *n* =335), which was conducted between 1 to 21 September 2016. During juvenile growth sessions, which occurred in August, October and November a total of 1,443 fish were caught during the sessions (Reach 4, *n*=33; Reach 3, *n*=870; Reach 2, *n*=540). River stage was too high to fish during the months of April, May, June, and July. Water temperatures were less than 5° C throughout the study area during the scheduled December fish growth sampling session which prohibited fish sampling according to the permit conditions. Consequently, winter juvenile growth data were not collected.

3.3.1 *2016 Seasonal Fish Size Index (Fish Growth) Results*

During 2016, a total of 2,264 fish were measured in all growth sessions. Rainbow trout made up most of the samples. A total of 236 chinook was caught and measured in total across 2016 (Reach 2, *n*=118; Reach 3, *n*=110; Reach 4, *n*=8). Chinook fry capture peaked with a total of 96 during September sampling, and dropped to 32 in November. No Age-1 chinook and coho were caught throughout the entire year. A total of 443 coho fry (Reach 4, *n*=17; Reach 3, *n*=235; Reach 2, *n*=191) and1585 rainbow trout (Reach 4, *n* =159; Reach 3, *n* =860; Reach 2, *n* =566) were caught in 2016, with most caught in September. Mean weights, standard deviation and total count, per species and age-class, by reach, are presented in Table 6.

Table 6: Mean fish weight (g), sample size (n) and standard deviation for each species, age-class of salmonids and for all reaches captured in the Lower Bridge River for growth information, August to November, 2016. The bold and italicized numbers indicate those species/age classes that were insufficient in achieving their target sampling size minimum threshold / reach

Species &	Sampling	Reach 2			Reach 3			Reach 4		
Age Class	Month	Mean	n	SD	Mean	n	SD	Mean	n	SD
CH - 0+	August	8.0	***31***	1.9	8.4	***17***	1.9	–	–	–
CH - 0+	September*	7.1	***49***	1.7	8.1	***39***	1.5	9.0	***8***	2.0
CH - 0+	October	8.7	***25***	2.0	9.0	***35***	2.1	–	–	–
CH - 0+	November	8.5	***13***	1.5	9.9	***19***	1.6	–	–	–
CO - 0+	August	4.6	60	1.7	3.8	***78***	1.4	–	–	–
CO - 0+	September*	4.9	***35***	1.4	4.6	***35***	1.3	5.2	***13***	1.1
CO - 0+	October	5.2	***44***	1.6	5.3	***80***	1.2	9.0	***3***	2.5
CO - 0+	November	5.6	***52***	1.3	5.9	***42***	1.3	9.7	***1***	–
RB - 0+	August	1.6	60	1.0	2.2	139	1.2	–	–	–
RB - 0+	September*	2.7	229	1.2	2.9	146	1.2	2.6	79	1.2
RB - 0+	October	2.8	60	1.3	3.6	***97***	1.1	–	–	–
RB - 0+	November	3.1	90	1.1	4.1	***42***	1.1	–	–	–
RB - 1	August	21.7	***18***	11.1	15.5	***82***	6.9	35.9	***2***	7.3
RB - 1	September*	19.8	***20***	6.5	16.5	***107***	6.1	18.0	47	8.4
RB - 1	October	19.5	***33***	8.3	18.2	131	7.6	44.0	***11***	13.0
RB - 1	November	16.8	***47***	9.0	21.7	***90***	15.9	31.1	***4***	7.3
RB - 2	August	53.7	***2***	15.5	92.3	***2***	22.1	73.6	***7***	24.6
RB - 2	September*	47.6	***2***	14.6	45.9	***8***	8.6	47.3	***4***	8.1
RB - 2	October	–	–	–	56.4	***7***	12.6	–	–	–
RB - 2	November	52.2	***5***	20.6	53.0	***9***	23.2	54.8	***5***	11.0

[a] Growth data for September were derived from fish sampled during the annual stock assessment.
(-) indicates that no fish were caught within that species and age-class.

Cells italicized and in bold (Table 6) demonstrate where the target number of fish per species and age-class (target of *n*=30 per site/reach; therefore, *n*=60 for Reach 2, *n*=120 Reach 3 and *n* =30 Reach 4) was not achieved for that reach. Overall, where sampling occurred and fish were caught in any of the fish/age categories across the reaches (*n* =20); results indicated that Reach 2 achieved targeted sampling 25% of the time, Reach 3 15% of the time and Reach 4, 10% of the time. More effort (electrofishing time) was put into electrofishing during the sessions in August, October and November, 2016 to increase the sample size to attempt to meet or exceed target numbers consistently in these specific months, however fish abundance likely limited catch. Due to insufficient sampling numbers, data were compiled; however, interpretation was limited.

3.3.2 *Standing Stock Assessment Results*

3.3.2.1 2016 Abundance Estimates and Trial Comparison

Total fish abundance across the reaches in 2016 was estimated at 80,120 fish (Table 7). Total juvenile rainbow (RB) numbered 59,240 fish and represented 74% of the total estimated abundance of fish (Table 7). Total 2016 coho fry (CO-0+) abundance was 10,050, or 13% of the total abundance in 2016. Total 2016 chinook (CH) fry abundance was estimated at 10,830 or 13% of the total (Table 7).

Rainbow fry numbered 39,500 and represented 49% of the total estimated abundance of juvenile fish in 2016. This value was lower than all of the Trial 1 and 2 estimates for this species/age class, and similar to estimates prior to the initiation of flow releases from Terzaghi Dam. Among the reaches, recent declines in rainbow fry abundance were apparent in Reach 2, Reach 3 and 4.

Table 7: 2016 abundance estimates for all species-age classes for reaches 2, 3 and 4. No error estimate was available for these data at the time of publication

	Abundance (Number of fish)		
Species-Age Class	**Reach 2**	**Reach 3**	**Reach 4**
CH - 0+	4,240	6,110	480
CO - 0+	3,610	5,340	1,100
RB - 0+	16,100	21,780	1,660
RB – 1+	2,080	15,570	2,050
Total	**26,030**	**48,800**	**5,290**

Total coho abundance was 10,500 or 12% of the total abundance in 2016, which was notably lower than estimates of chinook abundance in 2016. Coho numbers were lower than most of the abundances observed in the Pre-Flow period (mean = 25,000) (Table 8, Figure 14), and lower than all of the previous estimates during the years in Trial 1 (mean = 81,000) and Trial 2 (mean = 77,000). The majority of the decrease in abundance could be accounted for by changes in the estimates for reaches 3 and 4. CO fry abundance in Reach 2 was lower than the other Trial 2 estimates (with the exception of 2015), but similar to many of the Trial 1 and Pre-Flow estimates for that reach (Figure 14).

Chinook fry numbers were low relative to many of the earlier years, however the abundance appears to have stabilized across Trial 2. The 2016 estimate (n= 10,800) was similar to estimates during other recent years during Trial 2 (range = 10,000 to 14,000). By reach, the CH fry estimates were highest in Reach 3 and lowest in Reach 4 (Table 7.).

Rainbow parr (1+) abundance measured roughly 19,700 fish (or about 25% of the total 2016 abundance estimate). It should be noted that estimates of parr abundance tended to be more uncertain than for fry due to higher catch variability and reduced capture probability for this age class (Sneep and Korman, in prep). The issues related to catch variability and reduced capture probability made interpretations of RB1 abundance estimates more difficult, and interpretation of results should be cautious. The 2016 abundance value was similar to 2015 estimated abundance (19,000) and was lower than most of the other flow trial years. Much of this apparent reduction was potentially due to a decrease in abundance in Reach 3 and Reach 4, while abundance in Reach 2 appeared to increase

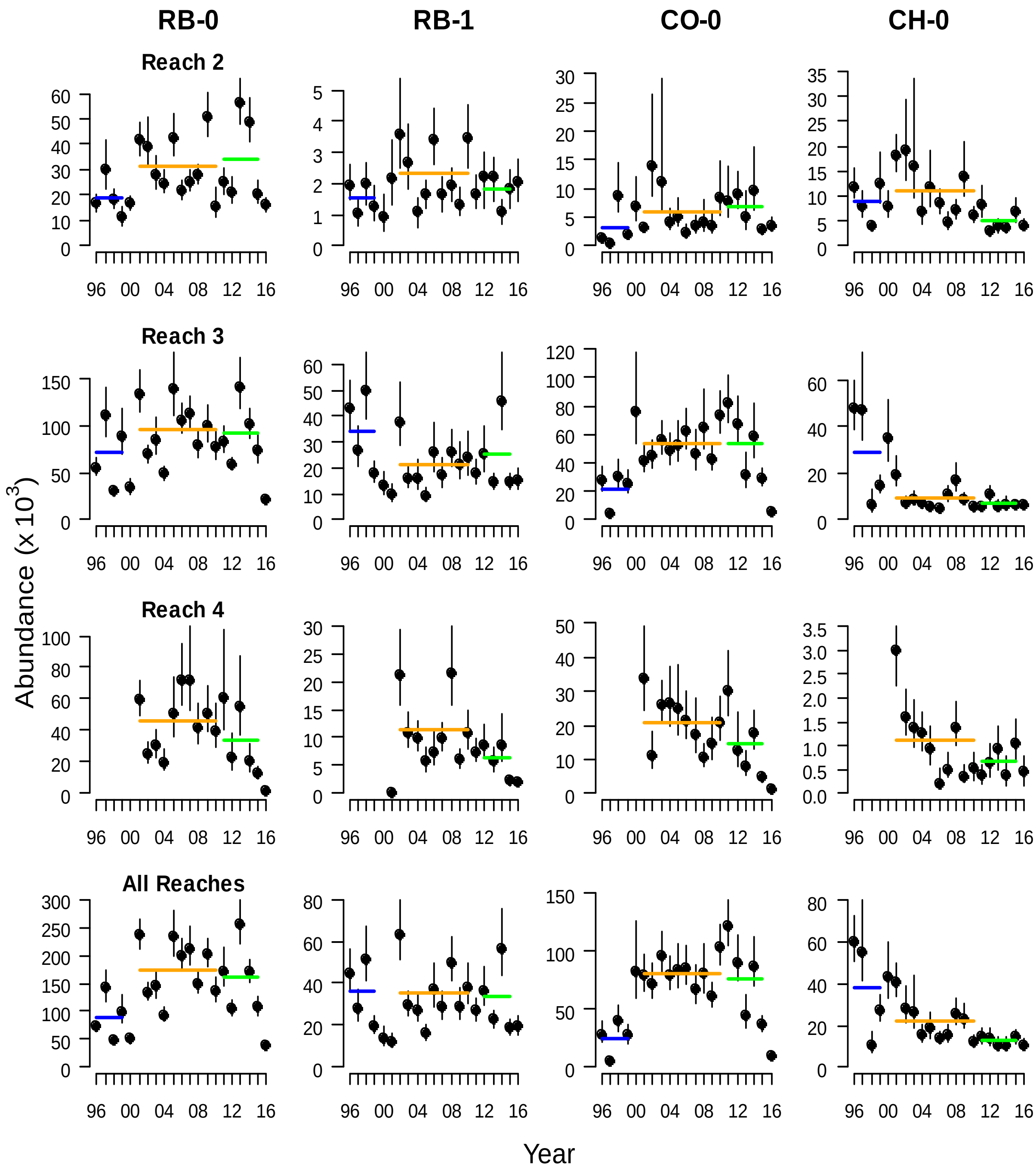


Figure 14: Annual median estimates of abundance (points) and 95% credible intervals across all reaches for RB-0, RB-1, CO-0, and CH-0. Horizontal lines show the average of annual median estimates across years for each flow trial period (blue=pre=flow, orange=trial 1, green=trial 2) (Korman, 2017)

The 2016 abundance estimate of approximately 80,000 fish was roughly 28% of the estimated mean abundance (286,000) during the years in Trial 2 (2011–2015); 26% of the average abundance during Trial 1 (313,000; 2000–2010); and 42% of the average estimated abundance during the Pre-Flow period (191,000; 1996–1999). Overall, the 2016 total abundance estimate was lower than any other annual estimate since monitoring was initiated, including all of the Pre-flow estimates before Reach 4 was wetted (Table 8).

Table 8: Estimated total abundance of salmonids in the LBR across reaches 2, 3 and 4 by flow treatment. No error estimate was available for these data at the time of publication

	Abundance (Number of fish in thousands)			
Species-Age Class	**Pre-Flow** (0 $m^3 \cdot s^{-1}$)	**Trial 1** (3 $m^3 \cdot s^{-1}$)	**Trial 2** (6 $m^3 \cdot s^{-1}$)	**2016** (22 $m^3 \cdot s^{-1}$)
CH - 0+	39	22	13	11
CO - 0+	25	81	77	10
RB - 0+	91	175	163	39
RB – 1+	36	34	33	20
Total	**191**	**313**	**286**	**80**

3.3.2.2 2016 Total Mean Biomass Estimates and Trial Comparison

Fish biomass across the reaches in 2016 was estimated at 605 kg (Table 9). Total RB biomass was 474 kg representing 78% of the total biomass (Table 9). Total 2016 CO0+ biomass was 48 kg, or 8% of the total biomass in 2016. Total 2016 CH0+ biomass was estimated at 84 kg or 14%.

Table 9: Estimated total mean biomass of salmonids in the Lower Bridge River (reaches 2, 3 and 4) during the standing stock assessment, September 2016

	Biomass (Kilograms)		
Species-Age Class	**Reach 2**	**Reach 3**	**Reach 4**
CH - 0+	30	50	4
CO - 0+	18	25	6
RB - 0+	47	69	6
RB – 1+	41	270	41
Total	**135**	**414**	**57**

Biomass was low across reaches 2, 3 and 4, compared to Trial 1 and Trial 2, while effects within Reach 2 were not as pronounced as the upper reaches (Table 9, Figure 15). Reach 3 biomass was estimated at 414 kg, which was the lowest level recorded since the start of the Pre-flow period for that reach (Table 10, Figure 15). Similarly, Reach 4 was also the lowest estimated biomass since the start of Trial 1, with a total value of 57 kg. CO fry biomass estimates were very low across all the reaches in 2016, (49 kg), which was approximately 80% less than the estimated biomass across the years in Trials 1 and 2. RB fry biomass was estimated at 122 kg, which was lower than average trial estimates from 1996 – 2015 during the Pre-Flow (mean = 249 kg), Trial 1 (mean = 305 kg) and Trial 2 (mean = 311 kg). Overall, CH fry biomass remained notably low at approximately 84 kg in 2016, which was comparable to the other Trial 2 years.

Similar to the abundance estimate, the 2016 biomass estimate of approximately 600 kg was also lower than any other annual estimate since the flow release was initiated, including all years within the Pre-flow estimates, before Reach 4 was wetted. In comparison, total mean biomass between 2011-2015 (Trial 2) was estimated at 1,223 kg; biomass between 2001 – 2010 (Trial 1) was 1,341 kg; and from 1996 – 1999 biomass was estimated at 1275 kg (Table 10; Figure 15).

Table 10: Estimated total mean biomass of salmonids in the LBR across 2, 3 and 4 by flow treatment. No error estimate was available for these data at the time of publication

	Biomass (Kilograms)			
Species-Age Class	**Pre-Flow** ($0\ m^3 \cdot s^{-1}$)	**Trial 1** ($3\ m^3 \cdot s^{-1}$)	**Trial 2** ($6\ m^3 \cdot s^{-1}$)	**2016** ($22\ m^3 \cdot s^{-1}$)
CH - 0+	228	134	92	84
CO - 0+	108	281	286	49
RB - 0+	249	305	311	122
RB – 1+	690	621	534	351
Total	**1275**	**1341**	**1223**	**606**

Populations of coho and rainbow fry in the LBR in 2016 may have been impacted by the hydrograph as abundance estimates were among the lowest since monitoring began in 1996 (Figure 14, Figure 15), although trends of declining abundance continued in reaches 3 and 4. Juvenile chinook populations were already very low and it is unclear if they were further impacted by the high 2016 flows. Similar patterns of apparent population declines for coho and rainbow fry were observed following previous spill scenarios within the LBR in 1997, which was an ~25 $m^3 \cdot s^{-1}$ spill, (Figure 14; McHugh, et. al 2015), following a spill in 1991 (Triton Environmental, 1992), and recently after a small spill event and stage fluctuation in 2015 which peaked around 20 $m^3 \cdot s^{-1}$ (McHugh, et. al 2015a). It is uncertain how the hydrographs contributed to the declines, and more data are needed at high flows to increase resolution of findings. The Discussion outlines several flow related factors that may have contributed to the low abundance and biomass estimates in 2016. Additional factors unrelated to higher flows from TRZ, such as a change in adult stock recruitment (spawner numbers across species) in the LBR, or elevated water temperatures during the fall and winter chinook egg incubation period, may have also influenced juvenile fish populations and are discussed in the following sections.

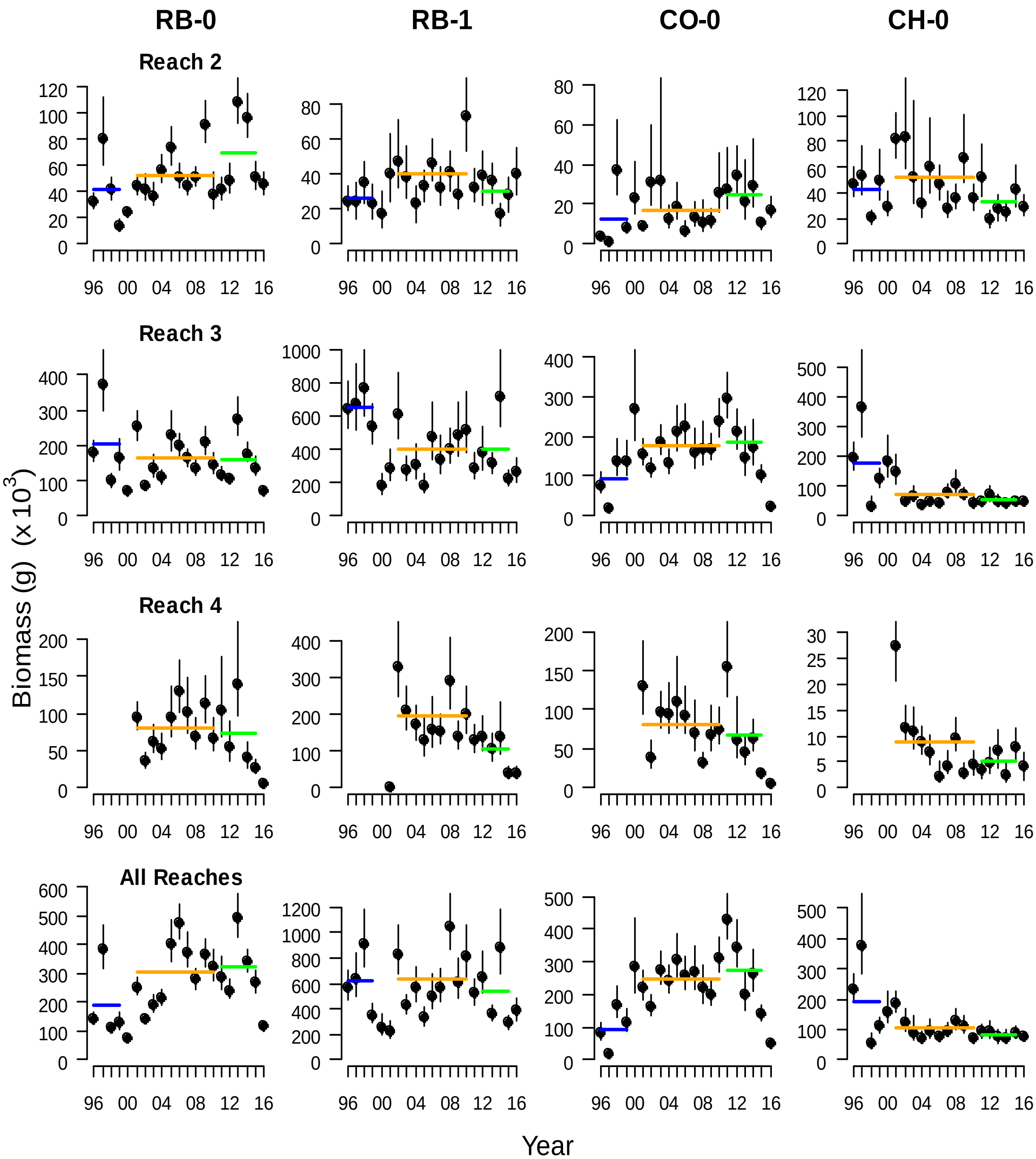


Figure 15: Annual median estimates of biomass (points) and 95% credible intervals across all reaches for RB-0, RB-1, CO-0, and CH-0. Horizontal lines show the average of annual median estimates across years for each flow trial period (blue=pre=flow, orange=trial 1, green=trial 2) (Korman, 2017)

3.3.2.3 LBR Stock and Recruitment Relationships

The relationship between escapement and juvenile production has been documented extensively in the literature for rivers where these parameters are well monitored. In order to make interpretations regarding the effects of the increased flows on juvenile abundance, it is assumed that escapement did not affect the number of juveniles present during sampling. This assumption required that escapement exceeded levels needed to fully seed the available habitat. As part of the LBR Synthesis Assessment (Sneep and Korman, in prep), the available stock recruitment information was assessed for CH and CO in Reach 3 and 4 to determine if abundance of juvenile salmon could have been influenced by LBR escapement during the flow trial periods. Data from each LBR flow trial period indicated relatively consistent fry production across a fairly broad range of adult escapement levels. In 12 out of 15 years where data were available, spawner density estimates for CO exceeded the numbers required to fully seed reaches 3 and 4 through Trial 1 and Trial 2 (Sneep and Korman, in prep). Analysis indicated that juvenile abundance estimates during Trial 1 and 2 (2000 – 2015) were likely not impacted by the number of parental spawners (up to and including 2014 escapements). For the 2015 brood year (2016 productivity) BCH found in their preliminary assessment that the 2016 productivity for chinook and coho was lower than expected after taking into consideration the effects of spawner abundance (Martins, 2017). This was more apparent during the 2015 brood year of coho, but was also been evident for chinook.

Stock and recruitment relationships for chinook have not been extensively documented in the literature. Based on the recommended DFO habitat seeding requirements of 51-80 spawners/km for mid Fraser River populations and LBR spawner estimates provided by BRGMON-3, the number of CH spawners may not have exceeded the DFO seeding recommendations since 2004. However, estimates for LBR CH and CO were likely biased low based on comparison with the more robust resistivity counter data available starting in 2014 (Sneep and Korman, in prep). Limitations regarding the historical reconstruction of LBR CH and CO escapement are discussed in more detail in the synthesis assessment. Accurate CH estimates from the resistivity counter in 2014 indicated that Reach 3 and 4 exceeded the DFO recommendation that year (with 63 spawners/ km). For this brood year in particular, no spawner limitation occurred that would have influenced juvenile production. However very few CH fry were captured during the standing stock assessment in the fall of 2015. These data demonstrated that the decline in CH fry abundance over Trial 1 and Trial 2 was not likely caused by adult escapement limitations alone (Sneep and Korman, in prep). Going forward, the collection of annual adult escapement data for CO and CH under BRGMON-3 will further support the development of LBR-specific stock-recruitment curves for these species.

3.3.2.4 Implications of Altered Thermal Regime for Chinook Emergence Timing

The thermal regime evident across Trials 1, 2 and 2016 (Figure 7) produced increased fall temperatures of approximately 2 – 4 C° during the CH egg incubation period, relative to the Pre-Flow baseline. This was documented within the LBR during Trial 1 to cause acceleration in the development of eggs and alevins, leading to early emergence of CH fry in the LBR (Sneep and Korman, in prep; Sneep and Hall, 2012). Coho and rainbow were likely not impacted by the elevated temperatures, as egg development occurs during a different season. While emergence timing is not likely determined by temperature alone, Table 11 (from Sneep and Korman, in prep) shows predicted dates of emergence, across the Pre-Flow, Trial 1 and Trial 2 periods, based on river temperature and accumulated thermal units in the LBR. Flows during the fall and winter were similar between Trial 1 and Trial 2; therefore predicted emergence dates were also similar.

Table 11: Predicted CH emergence date summary based on the Pre-Flow, Trial 1 and Trial 2 thermal regimes during the incubation period. Early to mid-winter emergence dates are highlighted in red; late winter dates are highlighted in yellow; and 'normal' dates are not highlighted (taken from Sneep and Korman, in prep)

Site	Pre-Flow		Trial 1 ($3\ m^3 \cdot s^{-1}$)		0 to 3 $m^3 \cdot s^{-1}$ Diff	Trial 2 ($6\ m^3 \cdot s^{-1}$)		0 to 6 $m^3 \cdot s^{-1}$ Diff
	Est. Emerge. Date	Incub. (# days)	Est. Emerge. Date	Incub. (# days)		Est. Emerge. Date	Incub. (# days)	
39.9			26-Nov	80		26-Nov	80	
36.5	15-Jan[a]	130	15-Dec	99	-31	16-Dec	100	-30
33.3	20-Apr	225	26-Jan	141	-84	2-Feb	148	-77
30.4	27-Apr	232	16-Feb	162	-70	2-Mar	176	-56
26.4	28-Apr	233	3-Mar	177	-56	18-Mar	192	-41
23.6	5-May	240	30-Mar	204	-36	25-Apr	230	-10
20.0	6-May	241	18-Apr	223	-18	24-Apr	229	-12

Survival was likely poor for fry that emerged in winter or early spring, and this was one of several probable causes of low fry abundance in reaches 3 and 4 following the initiation of the flow release from TRZ and through Trial 2 (Sneep and Korman, 2015). Three proposed fall flow alternatives ($1\ m^3 \cdot s^{-1}$; $0.5\ m^3 \cdot s^{-1}$ and $0.25\ m^3 \cdot s^{-1}$) were modelled during the LBR Synthesis Assessment to conceptualize the level of flows required to restore a more natural thermal regime. Thermal regime restoration was predicted to reduce early emergence and benefit CH survival (Sneep and Korman, 2015). A more detailed discussion regarding predicted CH emergence timing, subsequent survival, and results from the modelling exercise to restore the thermal regime are available in the LBR Synthesis Assessment (Sneep and Korman, 2015).

3.3.3 *Flow Rampdown Survey Results*

The scope of 2016 BRGMON-1 reporting on flow rampdown and stranding was limited to <15-$m^3 \cdot s^{-1}$, and data summarizing rampdown results for flows > 15 $m^3 \cdot s^{-1}$ can be found in the LBR Aquatic Monitoring (2016 High Flow) report (McHugh et al, 2017).

3.3.3.1 Terzaghi Dam Flow Release and River Stage Results

The initial rampdown transitioned the river from 15 $m^3 \cdot s^{-1}$ to 3 $m^3 \cdot s^{-1}$. In the subsequent September rampdown, flow from the lower-level outlet gates was reduced from 3 $m^3 \cdot s^{-1}$ to 1.5 $m^3 \cdot s^{-1}$ as per the Trial 2 WUP hydrograph. According to terms within the Bridge River Power Development Water Use Plan (2011), the maximum rates of stage change should not exceed 2.5 cm/hour or a total of 15 cm/day within Reach 4 of the Lower Bridge River. At the BCH compliance point, Rkm 36.8, the total maximum change per day limit (15 cm/day) was not exceeded during any of the 10 days (Table 12). Hourly stage change data were not available at the compliance point due to BC Hydro data logger malfunction. Table 12 summarizes the total changes in river stage elevation and the flow release volume at the estimated stage change compliance point for each ramping date in August and September. During the summer rampdown events, the relative stage data decreased across all sites in correspondence with the decrease in flow coming from the LLO gates at Terzaghi Dam. By being on site crews successfully salvaged fish regardless of hourly stage change rates.

Table 12: Stage change at the compliance point (Rkm 36.8) during each rampdown event

Rampdown Dates	TRZ $m^3 \cdot s^{-1}$ (Start)	TRZ $m^3 \cdot s^1$ (End)	Stage Change (cm)	Estimated stage change from 2016 compliance point rating curve*
05-Aug-16	15.3	13	7	6.8
08-Aug-16	13	11	6	6.8
09-Aug-16	11	9.3	6	6.3
10-Aug-16	9.3	7.7	7	6.7
17-Aug-16	7.7	6.4	6	6.2
18-Aug-16	6.4	5.1	7	7.1
23-Aug-16	5.1	4.1	6	6.3
24-Aug-16	4.1	3	9	8.3
27-Sep-16	3	2.2	9	7.4
28-Sep-16	2.2	1.5	6	8.1

*unverified stage-discharge rating curve

3.3.3.2 Stranding Risk

Due to morphological characteristics and predominately coarse in-stream substrate, the Lower Bridge River was sensitive to fish stranding. Stranding risk has historically been associated with the ramping rate, particularly within higher risk stage elevation ranges (Sneep, 2016, Crane Creek Enterprises, 2012). In general, the slower the river was ramped down, the lower the risk for adverse effects on fish. The cross-sectional channel shape was also influential; as the river volume dropped, the effect of each 1 cm flow reduction on river stage elevation increases.

Historically, flow ranges associated with the lowest fish stranding potential occurred between 15 and 9 $m^3 \cdot s^{-1}$. During stage reductions below 11 $m^3 \cdot s^{-1}$, the fish stranding impact increased and remained high for each subsequent ramping step (Sneep, 2016). In 2016, data followed similar patterns (Table 13). Stranding risk with the lowest stranding potential occurred between 15.3- $m^3 \cdot s^{-1}$ and 7.7 $m^3 \cdot s^{-1}$ and stranding risk increased and remained medium to high throughout the duration of the stage reductions (Table 13). Sites were ranked risk ranked according to the number of fish salvaged, per site, per stage reduction. Sites where the number of fish salvaged on a given day was > 100 were ranked as high risk, and color-coded red. Yellow cells represent where the number of fish captured was between 10 and 99; sites were ranked as medium risk. Low risk sites, where < 10 fish were captured were shaded green (Table 13).

Table 13: Strand-risk ratings for stranding sites on the Lower Bridge River based on the numbers of fish salvaged per site, during each stage reduction. Red = High Risk; Yellow = Moderate Risk; Green = Low Risk, as defined above

Salvage Site		41.0	39.6	Eagle	Bluenose	37.0	36.6	36.5	35.9	30.7	Russel Springs	Hell Bar	Muddy Hell	Micheal Moon	Grizzly Bar	House Rock
Ca. Area (m^2)		50	30	3900	140	525.0	150	60	680	120	210	625	280	500	2050	75
Start Flow	End Flow	Total Number of Salvaged Fish Per Site														
15.3	13			11	1				24							17
13	11			37												
11	9.3		18		34	38										
9.3	7.7			1	5				37			10				
7.7	6.4					19	48	24		27	29	58	111			
6.4	5.1						17	40	6		3	31	34		625	52
5.1	4.1					42	30								336	41
4.1	3				5	103					127				101	
3	2.2		29			46	9									
2.2	1.5	2	6			6	5						19	107		

3.3.3.3 Physical Habitat Attributes

Salvage was conducted at four new sites in Reach 3 and one new site in Reach 4 in 2016. Table 14 presents a summary of new fish salvage locations and their physical attributes. Crews were on site to implement salvage at all required sites from the protocol, as well as sites newly identified to pose a stranding risk when areas dewatered or isolated. Due to access issues and safety considerations related to high river stage, it was not possible to survey much of the river-right side of the channel on most of the August rampdown dates. Reach 2 and 1 were also not salvaged, as they are not included in the LBR Fish Stranding Protocol.

Table 14: Summary of site attributes for additional fish salvage locations in Reach 3 and 4 on the LBR during the rampdown in August and September, 2016

Reach	Rkm	Site Name	Bank	Area (m^2)	Description
4	41.0	41.0 Rkm	L	50	Large deep isolated pool
3	36.5	Rkm 36.5	L	60	Small isolated pool and side channel
3	30.7	Upper Russell	L	120	Side channel with boulders and pool
3	28.7	Below Hell	L	10	small side chanel with pool
3	27.5	Michael Moon	L	500	Medium side channel, with pools and riffles

3.3.3.4 Fish Salvage

The BCH LBR Fish Stranding Protocol (Sneep, 2016), which focuses on reaches 3 and 4 and omits reaches 1 and 2, guided the overall strategy for rampdown operations and monitoring in 2016. Fish salvage crews monitored the stranding and conducted salvage where necessary, over the duration of all of the rampdown events. Overall, data demonstrated a successful transition throughout the rampdown events on the LBR in 2016. Consequently, the majority of fish observed at identified salvage sites were successfully salvaged prior to stranding, Most of the fish salvaged during the rampdown event were rainbow and coho fry (Table 15). Most of these juvenile fish prefer shallow, grassy, protected habitat for rearing, and this habitat type is likely to dewater when flows are ramped down in the Lower Bridge River. Table 15 summarizes the number of fish salvaged by species for each day of the rampdown. Rainbow fry made up the majority of the fish salvaged, while coho made up most of the remainder. Table 16 summarizes the number of fish salvaged by date, type of activity (e.g. incidental “push” or active salvage), species and reach. In total, 2,371 fish were salvaged during all the rampdown events. Fish that were still in wetted habitat but were isolated from the main channel made up about 27% of all salvage types, with the majority of the remaining proportion being incidental capture (fish were occupying habitat that was still connected to the main flow, and were “pushed” or encouraged to vacate habitat areas that would isolate or dewater as the rampdown continued). Very few fish mortalities were observed (*n=12*), and only 89 fish were found stranded in dewatered habitat.

Table 15: Number of fish salvaged in reach 3 and 4, by species for each day of the rampdown, August and September, 2016

Species	Aug 5	Aug 8	Aug 9	Aug 10	Aug 17	Aug 18	Aug 23	Aug 24	Sep 27	Sep 28	TOTAL
Reaches 3, 4											
Chinook						12	2	3	4	9	30
Coho	20	33	4	3	57	154	81	53	10	25	440
Steelhead/RB	32	4	73	50	258	641	365	280	70	111	1884
Bull trout	1				1		1				3
Red Sided shiner			13			1					14
Total	53	37	90	53	316	808	449	336	84	145	2371

Table 16: Number of fish salvaged by reach, species and salvage category, August and September, 2016

Month	Reach	Species	Incidental	Isolated	Mortality	Stranded	Total
August	3	Bull Trout	1	1	-	-	2
		Chinook	10	4	-	-	14
		Coho	189	131	2	14	336
		Steelhead/RB	718	460	10	75	1263
		Red Sided Shiner	1	-	-	-	1
	4	Bull Trout	1	-	-	-	1
		Chinook	3	-	-	-	3
		Coho	68	1	-	-	69
		Steelhead/RB	426	14	-	-	440
		Red Sided Shiner	13	-	-	-	13
September	3	Bull Trout	-	-	-	-	0
		Chinook	9	-	-	-	9
		Coho	19	-	-	-	19
		Steelhead/RB	98	-	-	-	98
		Red Sided Shiner	-	-	-	-	0
	4	Bull Trout	-	-	-	-	0
		Chinook	2	2	-	-	4
		Coho	4	12	-	-	16
		Steelhead/RB	66	17	-	-	83
		Red Sided Shiner	-	-	-	-	0

4.0 DISCUSSION

4.1.1 *Answering the Management Questions and Current Challenges*

This report summarized data collected in implementation Year 5 for BRGMON-1 in the Bridge-Seton WUP. It also presents data from previous years and compares and contrasts data from separate trials wherever this is feasible. Data from this report will help to inform flow management decisions in the LBR.

The key relevant management questions, listed below, drive the program. They are intended to directly describe and reduce uncertainties about the effects of flow on the LBR aquatic ecosystem:

1) How does the in-stream flow regime alter the physical conditions in aquatic and riparian habitats of the Lower Bridge River ecosystem?
2) How do differences in physical conditions in aquatic habitat resulting from the in-stream flow regime influence community composition and productivity of primary and secondary producers in the Lower Bridge River?

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

Due to the nature of an adaptive management program such as the LBR and the importance of integrating new knowledge and information into assessments as time progresses, it is important to annually evaluate if the program is on track to answering these questions and address any challenges. Towards this effort, the discussion below attempts to summarize how the flow regime influenced the physical conditions and habitat, the primary and secondary benthic invertebrate response, and ultimately how these factors influenced the recruitment of juvenile fish populations in the LBR.

4.2 Question 1: How does the in-stream flow regime alter the physical conditions in aquatic and riparian habitats of the Lower Bridge River ecosystem?

4.2.1 *Thermal Regime*

Throughout 2016, Trial 1 and Trial 2, spring and fall temperatures were distinctly warmer, and summer temperatures were consistently cooler than observed in the Pre-Flow data. These effects were strongest in the upper reaches (Reach 3 and 4) and weakest in Reach 2, reflecting the primary influence of the hypolimnetic flow from TRZ. During higher flow periods, effects extended further downstream. The unregulated, Yalakom River flow helped buffer the impacts of the hypolimnetic flow release on the aquatic ecosystem primarily during WUP flows, and aided in thermal recovery and mitigation of impacts, particularly in Reach 2. However this effect appeared muted during peak flows of 2016.

Thermal regimes have distinct ecological relevance and differ in their variability, predictability of annual temperatures and monthly temperatures, and thermal events (the magnitude, frequency, duration time and rate of change in event). Fish and invertebrates are influenced by individual and interactive effects of flow and thermal modification (Olden and Naiman, 2010) and depend on certain temperatures as environmental cues, to complete their life cycle. In addition to physiological responses, behavioral responses have also been observed in other river systems. At elevated temperatures, Kuehne et al. (2012) found multiple and cumulative stressors changed juvenile behavior and these responses ultimately influenced development and reproduction, and the overall growth of organisms within the aquatic community. In the LBR, increased fall temperatures of approximately 2-4 C° relative to the Pre-Flow baseline influenced the reproduction of chinook over 2016, Trial 1 and Trial 2, and caused accelerated egg development and early emergence of fry during the winter months. Survival of fry that emerged early was likely low, particularly in reaches 3 and 4. A more natural thermal regime would mitigate this issue. This could potentially be achieved by a reduction in fall and winter flows from TRZ, or TRZ dam modifications that would facilitate releasing water with cooler temperatures. The elevated temperatures may not have accelerated coho egg development, but this requires more data to reduce this uncertainty.

Temperature data show that the program is on track to answering the management question. More data under higher flows in 2017 will provide further resolution of trends during high flows to help answer this question.

4.2.2 *Changes to Aquatic Habitat Depending on flow*

The largest benefits to aquatic habitat since the implementation of flow trials has been the rewetting of reach 4 and increasing the wetted widths in reach 2 and 3. High flows in 2016 increased the amount of wetted area in the river, however results indicated that cascade/rapid habitat became more numerous and prolific throughout reaches 2, 3 and 4. This may have reduced the suitability of rearing habitat as flow velocities were likely increased to above optimal thresholds throughout the reaches. Widespread cascade/rapid habitat may have also displaced fish from unsuitable habitat and inhibited the accessibility of potentially useable rearing habitat. In summary, increased velocity during high flows may have impacted juvenile fish behavior, migration or movement patterns and may help explain the decline in fish abundance and biomass observed across the reaches in 2016.

Data from this monitoring component currently aid in the understanding of how different flows influence aquatic habitat characteristics. To further answer this management question, future years of data collection would benefit from focusing on the habitat suitability during high flows, and validating and refining the predicted flow thresholds for rearing habitat.

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

4.3.1 *Primary producers' conclusion*

Periphyton accrual rates were similar between 2016, Trials 1 and 2. Differences in trends appear to be more closely associated with deposition and accumulation of nutrients from pink salmon spawning years in pink (odd years) and non-pink (even years) than flow regime. This natural trend may influence the availability of nutrients for juvenile growth.

4.3.2 *Secondary producers' conclusion*

The rewetting of Reach 4 significantly benefited the benthic invertebrate community in general by increasing total abundance and diversity within reaches 2, 3 and 4 of the LBR. During Trials 1 and 2, no significant differences were observed in response to flow changes. Higher flows under Trial 2 did not significantly increase the benthic invertebrate community abundance, or benefit the community composition. A decline in abundance was observed in 2016, but diversity remained high in the invertebrate communities. This may suggest habitat complexity may not have declined and possibly increased in Reach 2. Alternatively, the invertebrate communities could be in a state of change after the large flow increase in 2016 and require more time to find a new competition and predator prey equilibrium.

Data from these monitoring components show that the program is on track to answering this management question. However continued monitoring is recommended to improve resolution at the high flows.

4.4 **Question 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?**

Juvenile fish productivity increased by greater than 100,000 fish across Trial 1 and Trial 2, relative to the Pre-Flow period. This was mainly attributed to continuously wetting all of Reach 4 and increasing the wetted widths in reaches 2 and 3. In general, productivity changes in Trial 1 were similar to Trial 2. In other words, increases in flows under Trial 2 did not provide any additional benefit for juvenile production. Rainbow and coho fry both benefited significantly from the flow release in Trial 1 and Trial 2. In contrast, chinook production declined after the initiation of flow in 2000 and estimates were low, but similar across Trial 1 and Trial 2.

Total estimates of abundance and biomass of juvenile fish populations during high flows in 2016 were the lowest recorded since the monitoring program began. In 2016 rainbow fry and coho fry abundance and biomass estimates were among the lowest recorded values since monitoring began. Chinook fry populations remained low, but stable. The most notable patterns were observed (according to species) in reaches 3 and 4, which suggested that the higher peak flows in 2016 were likely a driving factor of the low estimates. Similar patterns were also observed in 2015 (Mchugh and Soverel, 2016).

While the long-term impact of high flows on fish productivity is uncertain at this time, several flow-related factors likely contributed to the decline in abundance and biomass within the study area in 2016. They included, but were not limited to: 1) juvenile fish were displaced from their normal habitat downstream or into different habitats outside of the areas sampled; 2) widespread water with high velocities likely limited habitat availability and use during the spring and summer rearing period; 3) unknown stranding risk throughout the LBR may have caused fish mortality in areas not actively monitored during fish salvage operations; and 4) successful reproduction for rainbows was potentially impeded by factors including migration challenges or habitat availability for spawners during high flows, or potential redd disturbance (scouring, smothering, or dewatering during high flows and subsequent stage reduction). Additional factors unrelated to higher flows from TRZ, such as a change in adult stock recruitment (spawner numbers across species) in the LBR, or elevated water temperatures during the fall and winter egg incubation period may have also influenced juvenile fish populations. Some of these factors may also help to explain similar declines observed in 2015. More years of data collection under higher flows as well as the collection of annual adult escapement data under BRGMON-3 will reduce this uncertainty.

Data demonstrate the program is on track to answering this management question. Since the abundance and biomass of target species and age classes were reduced following increased peak flows in 2015 and 2016, the relationship between higher flows and juvenile salmon production should continue to be monitored in the LBR. The adaptive management of the LBR would also benefit from more refined information regarding the relationship between optimal flow thresholds and 1) rearing habitat quantity and quality, and 2) substrate movement, egg development and emergence timing.

4.5 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 $m^3 \cdot s^{-1}$?

The LBR Fish Stranding Protocol (Sneep, 2016) was effective in guiding the overall strategy and facilitating the rampdown in 2016 in Reach 3 and 4, and mainly on river left. Fish salvage site results from 2016 illustrated the dynamic nature of the riverbed following high flows; several sites were actively salvaged, where historical salvage was not conducted and stranding was not observed. Flow ranges associated with the lowest fish stranding potential occurred between 15 and 7.7 $m^3 \cdot s^{-1}$. Based on available data, Sneep (2016) determined that at flows below 9 $m^3 \cdot s^{-1}$, the fish stranding impact increased and remained high for each subsequent ramping step (Sneep, 2016). Data from 2016 support these conclusions and show that the program is on track to answering this management question, in the specific and limited geographic area that is the current focus of the protocol. However continued monitoring is recommended to improve resolution of savage results at high flows and expand the geographic focus of the salvage effort.

4.5.1 *Future Research and Monitoring*

Ecosystems are complex, and have multiple interactive and cumulative factors and linkages. Adaptive management success is predicated on being able to accurately predict the response of the aquatic ecosystem to flow changes. This report provides information regarding the predicted and observed benefits and ecosystem response to the instream flow release. However, uncertainties still confound questions regarding the long-term ecological benefits and costs from the release of instream flow from Carpenter Reservoir, and the effects on the aquatic productivity of the Lower Bridge River ecosystem. More years of data collection will continue to reduce this uncertainty.

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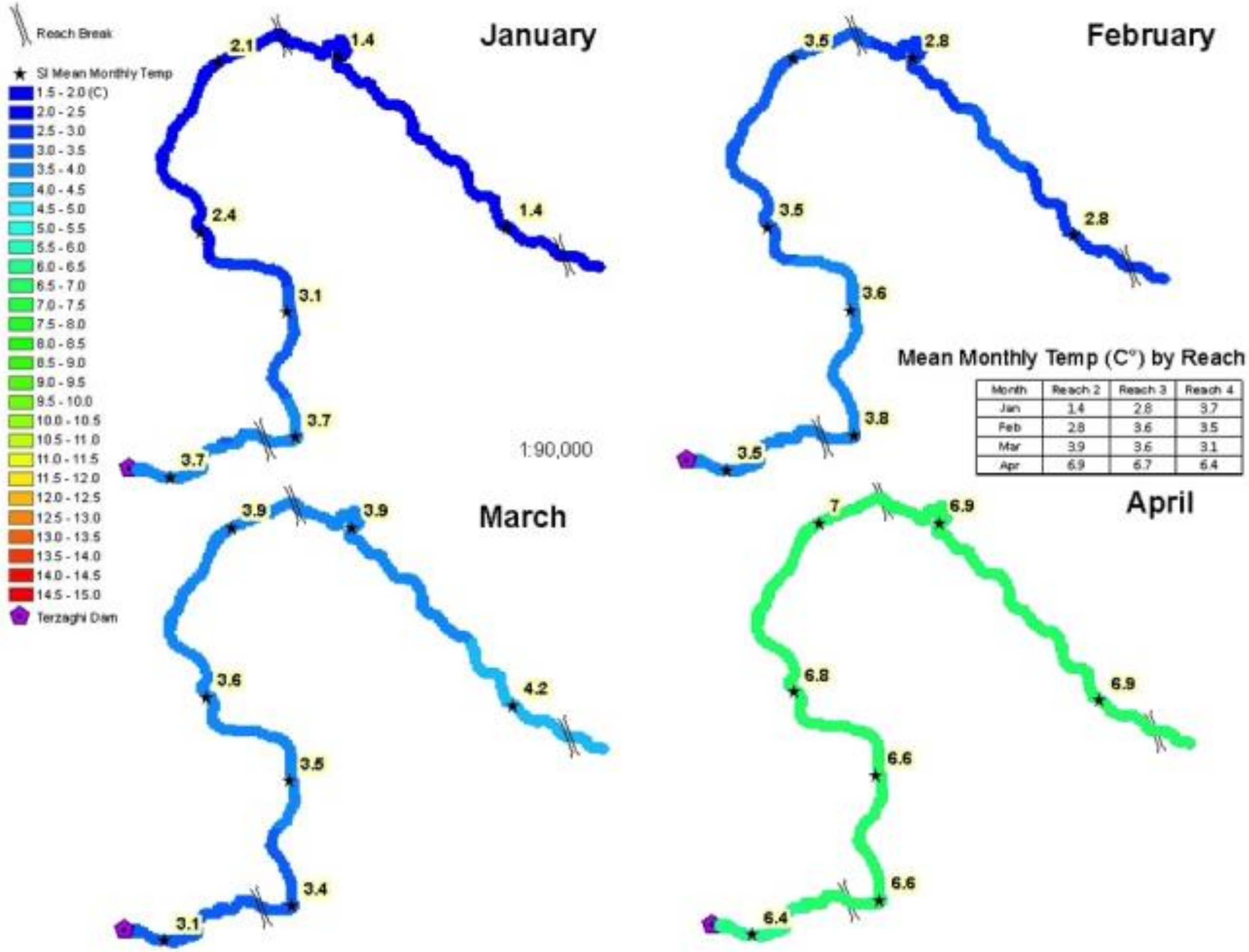
7.0 SUMMARY COST

Table 17: Summary Cost Table: Costs per study are shown as a total per year including inflation and contingency

Lower Bridge River Aquatic Monitoring	***2016***
BRGMon-1	Implementation Yr 5
Total cost	$213,371.00

8.0 APPENDIX A

8.1 Additional Tables and Figures



Month	Reach 2	Reach 3	Reach 4
Jan	1.4	2.8	3.7
Feb	2.8	3.6	3.5
Mar	3.9	3.6	3.1
Apr	6.9	6.7	6.4

Figure 16: **Temperature schematic of mean monthly water temperature (C°) recorded at each site index location along the LBR in January, February, March and April of 2016. Site indices on map in order from upstream to downstream (Rkm): 39.9, 36.5, 33.3, 30.4, 26.4, The colour ramp represents warmest water temperatures with shades of red and decreasing water temperatures progressing into orange and yellow, followed by green and finally the dark blue colour representing the coldest temperatures**

Table 18: Median count per basket samples for the five most abundant invertebrate taxa in each reach (ranks 1 through 5) taken from Stamford (2017)

Reach	Trial	Chironomidae (median)	Hydropsychidae (median)	Simuliidae (median)	Baetidae (median)	Ephemerellidae (median)	Heptageniidae (median)
2	0 CMS	33	51	2	28	8	33
	3 CMS	1457	196	186	223	103	277
	6 CMS	1485	171	47	82	130	312
	22 CMS	75	10	14	98	51	171
3	0 CMS	318	13	116	274	7	51
	3 CMS	396	42	72	488	102	292
	6 CMS	624	7	44	277	177	452
	22 CMS	128	1	4	100	40	219
4	0 CMS	0	0	0	0	0	0
	3 CMS	946	33	14	467	59	201
	6 CMS	815	61	1	686	599	113
	22 CMS	364	32	1	12	266	5

9.0 **DISCLAIMER**

No environmental assessment can wholly eliminate uncertainty regarding the potential for unrecognized environmental conditions in connection with water, land or property. Any use that a third party makes of this report, or any reliance on decisions made based on it, is the responsibility of such third parties. Coldstream Ecology, Ltd. accepts no responsibility for damages, if any suffered by any third party because of decisions made or actions based on this report. No other warranty, expressed or implied, is made.

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