Columbia River Project Water Use Plan

Lower Columbia River

Implementation Year 9

Reference: CLBMON#44

Lower Columbia River Physical Habitat and Ecological Productivity Monitoring

Study Period: 2016

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Lower Columbia River Physical Habitat and Ecological Productivity Monitoring (Year 9)

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ACRONYMS AND ABBREVIATIONS

AFDW	ash free dry weight
AICc	Akaike information criterion corrected for small sample sizes
ALR	Arrow Lakes Reservoir
ANCOVA	Analysis of covariance
BBK	Birchbank
BC Hydro	British Columbia Hydro and Power Authority
BRD	Combined discharge from Brilliant Dam, including spill and the Brilliant Dam expansion project
CART	Classification and Regression Tree
Caro Labs	Caro Environmental Laboratories (Kelowna, B.C.)
Celgar	Zellstoff Celgar Mill
CFU	colony forming unit
chl-a	Chlorophyll-a
CRIEMP	Columbia River Integrated Environmental Monitoring Program
CONT	continuation of MWF/RBT flows (2008-2016).
Didymo	*Didymosphenia geminate*
DIN	Dissolved Inorganic Nitrogen
EPT	Ephemeroptera (mayflies), Plecoptera (stoneflies), Trichoptera (caddisflies)
FFF	fall fluctuating flow
HBI	Hilsenhoff Biotic Index
HLK	Hugh L. Keenleyside
GAM	Generalized Additive Model
QA/QC	Quality assurance, quality control
km	kilometer
L	litre
LCR	Lower Columbia River
m	metre
m asl	metres above sea level
max	maximum value
MCR	Middle Columbia River
min	minimum value
MWF	Mountain Whitefish
n	sample size
NMDS	Non metric multidimensional scaling
NTU	nephelometric turbidity units
PCA	principal component analysis
PERMANOVA	permutational multivariate analysis of variance
POM	particulate organic material
RBT	Rainbow Trout
RVI	relative variable importance
SD	standard deviation
STD	standardized
T-P	total phosphorus
WQIS	water quality index station
UTM	Universal Transverse Mercator
WUP CC	Columbia River Water Use Plan Consultative Committee

DEFINITIONS

The following terms are defined as they are used in this report.

Term	Definition
Accrual rate	A function of cell settlement, actual growth and losses (grazing, sloughing)
Algae bloom	A superabundant growth of algae
Anaerobic/anoxic	Devoid of oxygen
Autotrophic	An organism capable of synthesizing its own food from inorganic substances, using light or chemical energy
Benthic	Organisms that dwell in or are associated with the sediments
Benthic production	The production within the benthos originating from both periphyton and benthic invertebrates
Catastrophic flow	Flow events that have population level consequences of >50% mortality
Cyanobacteria	Bacteria-like algae having cyanochrome as the main photosynthetic pigment
Diatoms	Algae that have hard, silica-based "shells" frustules
Diel	Denoting or involving a period of 24 hours
Epilithic algae	Algae that grow on hard inert substrates, such as gravel, cobbles, boulders
Eutrophic	Nutrient-rich, biologically productive water body
Flow	The instantaneous volume of water flowing at any given time (e.g.1200 m^3/s)
Freshet	The flood of a river from melted snow in the spring
Functional Feeding group	(FFG) Benthic invertebrates can be classified by mechanism by which they forage, referred to as functional feeding or foraging groups
Heteroscedasticity	Literally “differing variance”, where variability is unequal across the range of a second variable that predicts it, from errors or sub-population differences.
Heterotrophic	An organism that cannot synthesize its own food and is dependent on complex organic substances for nutrition.
Laminar	Non-turbulent flow of water in parallel layers near a boundary
Light attenuation	Reduction of sunlight strength during transmission through water
Limitation, nutrient	A nutrient can limit or control the potential growth of organisms e.g. P or N
Linear Regression Model	Linear regression attempts to model the relationship between two variables by fitting a linear equation to observed data
Macroinvertebrate	An invertebrate that is large enough to be seen without a microscope
Mainstem	The primary downstream segment of a river, as contrasted to its tributaries
Mesotrophic	A body of water with moderate nutrient concentrations
Microflora	The sum of algae, bacteria, fungi, *Actinomycetes,* etc., in water or biofilms
Morphology, river	The study of channel pattern and geometry at several points along a river
Peak biomass	The highest density, biovolume or chl-a attained in a set time on a substrate
Periphyton	Microflora that are attached to aquatic plants or solid substrates
Phytoplankton	Algae that float, drift or swim in water columns of reservoirs and lakes
Ramping of flows	A progressive change of discharge into a stream or river channel
Redd	A spawning nest made by a fish, especially a salmon or trout
Riffle	A stretch of choppy water in a river caused by a shoal or sandbar
Riparian	The interface between land and a stream or lake
Salmonid	Pertaining to the family *Salmonidae*, including the salmons, trouts, chars, and whitefishes.
Substrates	Substrate (sediment) is the material (boulder cobble sand silt clay) on the bottom of a stream.
Taxa Taxon	A taxonomic group(s) of any rank, such as a species, family, or class.
Thalweg	A line connecting the lowest points of a river, usually has the fastest flows
Zooplankton	Minute animals that graze algae, bacteria and detritus in water bodies

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EXECUTIVE SUMMARY

This is a multi-year study of physical habitat and ecological productivity on the Lower Columbia River (LCR) between the outflow of the Hugh L. Keenleyside Dam and the Birchbank gauging station. The aim of the study is to address management questions and hypotheses that examine the influence of three different managed flow periods (Mountain Whitefish (MWF) Jan 1 - Mar 31; Rainbow Trout (RBT) Apr 1 - Jun 30; and fall fluctuating (FFF) Sep 1 - Oct 31) on select physical habitat and ecological productivity measures. Table 0-1 summarizes the management questions, hypotheses and results to date.

In 2016, LCR flows originated from Hugh L. Keenleyside Dam (50.9 %), Kootenay River (45.5%) and smaller contributing tributaries. Freshet flows in 2016 were lower and occurred earlier when compared to the previous five years (2011-2015) of the study. Regression modelling of recorded river elevations and flows were used to predict river elevations during pre, post and continuous MWF and RBT flow periods. The river level difference between MWF maximum peak spawning and minimum incubation was greater before MWF flows were implemented than they were with post and continuous MWF flows. Similarly, cumulative elevation drops that occurred during pre-RBT flows were significantly higher than those determined during post and continuous RBT flow periods.

Water temperatures varied seasonally, and ranged from ~3.1 to 19.5°C in 2016. Regression modeling during each of the flow periods of cumulative data to date indicated that the influence of flow on LCR water temperature was relatively weak compared to other model predictors such as air temperature and reservoir temperature.

Water quality sampling was not conducted in 2016 and is not planned for the remainder of the contract. Previously (2008-2015), a suite of water quality parameters were collected on four occasions annually and they generally indicated good water quality in both the Kootenay River and LCR. The water quality sampling regime was suspended because the point sampling did not provide enough data to statistically assess the potential effects of the three managed flow periods on the water quality of LCR. The baseline water quality data and other lines of evidence have led us to conclude that the managed flow periods (MWF, RBT and FFF) have minimal to no effect on water quality compared to other factors such as freshet, anthropogenic nutrient donation, groundwater inputs, and even photosynthesis.

In 2016, numerous periphyton and benthic invertebrate productivity metrics, including total biovolume/biomass, chl-a and total abundance, were quantified from sampling during the winter, summer and fall using artificial substrate samplers deployed from 0 to 6 m depths for 10 weeks. Data was combined and analyzed with previous years to reveal that periphyton and benthic invertebrate communities were productive, diverse and variable. The benthic community composition during the winter was taxonomically unique compared to summer and fall. For example, Trichoptera (net-spinning caddisflies) were the dominant group in the summer and fall, while Gastropoda and Diptera dominated winter samples. Likewise, Didymo contributed to high winter biovolumes, compared to summer and fall. Most production metrics were comparable to those from other large, moderately productive rivers. The cumulative data to date suggested that periphyton and invertebrate production varied both seasonally and annually.

Periphyton production was greatest during the winter, followed by the fall and summer. The periphyton community was dominated by diatoms. Accrual data indicated that peak periphyton biomass was reached in 6-7 weeks in summer, after more than 8 weeks in the fall, and in greater than 10 weeks during winter. Cool, stable, lower velocity winter flows appeared to favor the growth of mats of the nuisance diatom, Didymo. Donations of algae from Arrow Lake Reservoir made significant contributions to LCR in the summer and fall, but not during the winter. Predictive statistical modelling was used to test if MWF, RBT and FFF resulted in an

increase of total biomass accruals. The predictive variables included velocity, substrates and flow variability that was specifically associated with each managed flow period. The models ranged in their predictive capabilities. They generally explained little to moderate levels of variation and therefore were not adequate to infer causation. The summer and fall models were more effective than winter models, likely because they included six years of data compared to three years of data for the winter. The results to date indicate that the managed flow periods had variable effects on periphyton production. During the MWF flow period, increased flow variability was associated with increased chl-a, while periphyton production was not positively influenced during the RBT and FFF periods.

The stable LCR flows appeared to benefit the benthic invertebrate community, not only in abundance, but also in the prevalence of more sensitive, high-quality fish food taxa such as Trichoptera. Very low fall flows in 2016 caused desiccation of shallow substrates and minimal benthic invertebrate presence following desiccation. Trichoptera (net-spinning caddisflies) were the dominant group in the summer and fall, while Gastropoda and Diptera dominated winter samples. The seasonal shift in the benthic invertebrate community composition is likely a natural shift.

Predictive statistical modelling was used to test if MWF, RBT and FFF had an effect on benthic invertebrate community composition, biomass, and abundance. Like the periphyton models, the models ranged in predictive capabilities and therefore cannot be used to elucidate casual relationships. During the MWF flow period, models suggested increased flow variability had an effect on benthic community composition, specifically diversity and richness. However, models did not detect an effect of flow variability on benthic invertebrate abundance and biomass during the MWF flow period. The effects of freshet overshadowed the effects of flow variability associated with the RBT flow period on benthic invertebrate community composition, biomass, and abundance. During the FFF period, abundance was the only benthic invertebrate metric that was effected by flow variability.

The two metrics of percent EPT and percent quality forage (EPT+Dipteran) were used to test if MWF, RBT, and FFF increase the availability of fish food organisms. To confirm the suitability of these indices, fish stomach contents from RBT and MWF caught in fall 2012 and 2014 were analyzed. The fish stomachs were primarily composed of Trichoptera and some Diptera, which confirms that percent quality forage is likely the best index to assess the availability of fish food organisms. Modelling results suggested that increased flow variability had a positive effect on the availability of fish food (measured as percent quality forage) during the FFF period only.

The literature clearly demonstrates that variables such as flow, velocity and substrates play a role in the overall characterization of the benthic community; this was further supported through our predictive modelling. However, the challenge occurs when trying to tease apart general flow variability from that of the MWF, RBT and FFF managed flows. To do this, we have attempted to develop predictive flow variables that are specific to each flow period. But the problem is that specific flow variables are highly correlated with overall flow and therefore it is impossible to determine if it is truly the managed flow that is having an effect and not just flow variability in general. To date, the data seems to indicate that when flows are high (e.g. during freshet), the effects of the managed flow period (RBT) is nominal, however in the fall and winter (FFF and MWF) when the flows are more stable, then the managed flow periods appear to play a larger role in shaping the overall benthic community. The goal of the remaining three years of this contract is to further explore better ways to isolate the true effects of each managed flow period from overall flow variability and to further improve our confidence in the roles of each managed flow period and how they may affect the individual metrics of the benthic community.

Table 0-1: CLBMON-44 Status of Objectives, Management Questions and Hypotheses After Year 9

Management Questions	Management Hypotheses	Year 9 (2016) Status
Physical Habitat Monitoring Q.1. How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect water temperature in LCR? What is the temporal scale (diel, seasonal) of water temperature changes? Are there spatial differences in the pattern of water temperature response?	Ho1phy: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the seasonal water temperatures regime of LCR.	The influence of flow on LCR water temperature was relatively small compared to other factors, and was negatively associated with river temperature during all flow periods. LCR water temperatures were most strongly correlated with air temperature and reservoir water temperature in all flow periods. Based on all analyses to date, flow is not an important determinant of river temperature (Scofield *et al.* 2011; Olson-Russello *et al.* 2015). Given the small influence of flow on LCR water temperature, the **null hypothesis is tentatively accepted.**
Physical Habitat Monitoring Q.2. How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect the seasonal and inter-annual range and variability in river level fluctuation in LCR?	Ho2phy: Continued implementation of MWF and RBT flows does not affect seasonal water levels in LCR. Ho2Aphy: Continued implementation of MWF flows does not reduce the river level difference between the maximum peak spawning flow (1 Jan to 21 Jan) and the minimum incubation flow (21 Jan to 31 Mar). Ho2Bphy: Continued implementation of RBT flows does not maintain constant water level elevations at Norns Creek fan between 1 Apr and 30 Jun.	Analyses suggest that river flow is an important determinant of water levels. At all locations, the river level difference between MWF maximum peak spawning and minimum incubation was greater during pre-MWF flows than during post and continuous MWF flows. Similarly, river elevation data from monitoring stations WQIS2 and WQIS3 were regressed with flow data. For both stations, the cumulative elevation drops that occurred during pre-RBT flows (1984-1991) were significantly higher than those determined during post (1992-2007) and continuous (2008-2016) flow periods. We therefore **reject all three** (Ho2phy, Ho2Aphy, Ho2Bphy) null hypotheses.
Physical Habitat Monitoring Q.3. How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect electrochemistry and biologically active nutrients in LCR?	Ho3phy: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the water quality of LCR. Ho3Aphy: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the electrochemistry of LCR.	There was no further testing of these hypotheses in 2016, as water quality sampling was not conducted. Analysis from previous years suggested that the influence of the managed fish flows on water quality is subtle compared to the stronger effects such as freshet, anthropogenic nutrient donation, groundwater inputs, and even photosynthesis within LCR. The potential effects of nutrient donation from the Arrow Lakes Nutrient Restoration Program on LCR water quality was explored in 2016. Although the analysis is preliminary, it indicated that nutrient donation may affect dissolved inorganic nitrogen (DIN) within LCR. We therefore continue to tentatively **accept the null hypotheses** HO_{3phy}, HO_{3Aphy}, and HO_{3Bphy} and assume that managed fish flows, whether they be MWF, RBT or FF flows, have no effect on the water quality of LCR.

	Ho3Bphy: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the availability of biologically active nutrients of LCR.	
Ecological Productivity Monitoring Q.1. What are the composition, abundance, and biomass of epilithic algae and benthic invertebrates in LCR?	Ho1: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not affect the biomass, abundance and composition of benthic invertebrates in LCR.	We **tentatively reject Ho1Aeco** as statistical models suggest flow variability has a minor effect on benthic invertebrate community composition (diversity and species richness) during the MWF flow period. . Results for this flow period are preliminary as there is only three years of data and large differences in operations between the study years could be amplifying the effects of variable flows on benthic invertebrate community metrics. We **tentatively accept Ho1Beco** because modeling suggests there is no detectable effect of managed flow variability on benthic invertebrate biomass, abundance and community composition during the RBT flow period. We suspect the reason that the effects of managed flow is un-detectable, is due to the higher flows and greater flow variability that occur during the freshet period. The higher flows essentially swamp out any effect of the RBT flow period on benthic invertebrate community metrics. We **tentatively reject Ho1Ceco** because models suggest flow variability has a minor effect on benthic invertebrate abundance during the FFF period.
	Ho1Aeco: Continued implementation of MWF does not affect the biomass, abundance and composition of benthic invertebrates in LCR.	
	Ho1Beco: Continued implementation of RBT flows does not affect the biomass, abundance and composition of benthic invertebrates in LCR.	
	Ho1Ceco: Continued fluctuations of flow during the fall do not affect the biomass, abundance and composition of benthic invertebrates in LCR.	
Ecological Productivity Monitoring Q.2. What is the influence of MWF and RBT flows during winter and spring, and fluctuating flows during fall on the abundance, diversity, and biomass of benthic invertebrates?	Ho2eco: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not increase total biomass accrual of periphyton in LCR.	For Ho2Aeco we have equated total biomass accrual to the periphyton production metrics of chl-a, biovolume and abundance. We **tentatively reject Ho2Aeco** because statistical models suggest that an increase in managed flow variability during the MWF flow period is correlated with an increase in periphyton production (chl-a). Results for biovolume could not be interpreted because the model explained limited variation. We **tentatively accept Ho2Beco** because there is no detectable effect of managed flow variability on periphyton production during the RBT flow period. This is likely due to the same reasons described for benthic invertebrates. We **tentatively accept Ho2Ceco** because the models suggested that increased flow variability during the FFF period resulted in a decrease of chl-a and biovolume.
	Ho2Aeco: Continued implementation of MWF does not increase total biomass accrual of periphyton in LCR.	
	Ho2Beco: Continued implementation of RBT flows does not increase total biomass accrual of periphyton in LCR.	
	Ho2Ceco: Continued fluctuations of flow during the fall do not increase total biomass accrual of periphyton in LCR.	
Ecological Productivity Monitoring Q.3. Are organisms that are used as food by juvenile and adult MWF and RBT in LCR supported by benthic production in LCR?	Ho3eco: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not increase the availability of fish food, organisms in LCR	We **tentatively accept Ho3Aeco** because models suggest percent quality forage is not influenced by flow variability during MWF flow period. Results for percent EPT could not be interpreted because the model explained such limited variation. We **tentatively accept Ho3Beco** because models suggest flow variability during RBT flow period did not increase the availability of fish food organisms (percent EPT and percent quality forage). We **tentatively reject Ho3Ceco** because models suggest an increase
	Ho3Aeco: Continued implementation of MWF flows does not increase availability of fish food organisms in LCR.	
	Ho3Beco: Continued implementation of RBT flows does not increase availability of fish food organisms in LCR.	

	Ho_{3Ceco}: Continued fluctuations of flows during the fall do not increase availability of fish food organisms in LCR.	in flow variability during FFF period increased the availability of fish food organisms (percent EPT and percent quality forage). Analysis of fish stomach contents confirmed RBT and MWF primarily consume Trichoptera (caddisfly) and Simuliidae (blackfly), both of these are included in percent quality forage.

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Appendix A – Supplemental Results

1.0 INTRODUCTION

This is a multi-year study of the physical habitat and ecological productivity on the Lower Columbia River (LCR), between the outflow of the Hugh L. Keenleyside (HLK) Dam and the Birchbank (BBK) gauging station. Over the past decade, BC Hydro and Power Authority (BC Hydro) has altered operations of HLK Dam to minimize the impacts of winter and early summer flows on fish spawning and rearing habitats in LCR.

This study aims to examine the influence of the regulated winter and early summer flow periods, compared to fluctuating flows in the fall, on select physical habitat and ecological productivity measures. This report addresses Year 9 (2016) of the study and includes both historic and 2016 data pertaining to the hydrology and benthic productivity of LCR.

1.1 Management Questions

The Columbia River Water Use Plan Consultative Committee (WUP CC) generated a set of management questions and hypotheses that relate to three different flow periods including:

1) Mountain Whitefish (MWF) spawning (Jan 1 – Jan 21) and incubation (Jan 22 – Mar 31). The purpose of the MWF flow period is to reduce the difference between peak flows during spawning and minimum flows during egg incubation;
2) Rainbow Trout (RBT) protection flows (Apr 1 – Jun 30). The purpose of this flow period is to reduce water elevation drops during the RBT spawning period; and
3) Fall fluctuating flow (FFF) (Sep 1 – Oct 31). This period is used to provide background data outside of regulated RBT and MWF flows.

The management questions addressed by the physical habitat and ecological productivity monitoring programs are (BC Hydro 2007):

Physical Habitat Monitoring

1) How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect water temperature in LCR? What is the temporal scale (diel, seasonal) of water temperature changes? Are there spatial differences in the pattern of water temperature response?
2) How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect the seasonal and inter-annual range and variability in river level fluctuation in LCR?
3) How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect electrochemistry and biologically active nutrients in LCR?

Ecological Productivity Monitoring

What are the composition, abundance, and biomass of epilithic algae and benthic invertebrates in LCR?

1) What is the influence of the MWF and RBT flows during winter and spring, and fluctuating flows during fall on the abundance, diversity, and biomass of benthic invertebrates?

2) Are organisms that are used as food by juvenile and adult MWF and RBT in LCR supported by benthic production in LCR?

1.2 Management Hypotheses

Physical Habitat Monitoring

HO_{1phy}: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the seasonal water temperatures regime of LCR.

HO_{2phy}: Continued implementation of MWF and RBT flows does not affect seasonal water levels in LCR.

HO_{2Aphy}: Continued implementation of MWF flows does not reduce the river level difference between the maximum peak spawning flow (1 Jan to 21 Jan) and the minimum incubation flow (21 Jan to 31 Mar).

HO_{2Bphy}: Continued implementation of RBT flows does not maintain constant water level elevations at Norns Creek fan between 1 Apr and 30 Jun.

HO_{3phy}: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the water quality of LCR.

HO_{3Aphy}: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the electrochemistry of LCR.

HO_{3Bphy}: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the availability of biologically active nutrients of LCR.

Ecological Productivity Monitoring

HO_{1eco}: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not affect the biomass, abundance and composition of benthic invertebrates in LCR.

HO_{1Aeco}: Continued implementation of MWF does not affect the biomass, abundance and composition of benthic invertebrates in LCR.

HO_{1Beco}: Continued implementation of RBT flows does not affect the biomass, abundance and composition of benthic invertebrates in LCR.

HO_{1Ceco}: Continued fluctuations of flow during the fall do not affect the biomass, abundance and composition of benthic invertebrates in LCR.

HO_{2eco}: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not increase total biomass accrual of periphyton in LCR.

HO_{2Aeco}: Continued implementation of MWF does not increase total biomass accrual of periphyton in LCR.

HO_{2Beco}: Continued implementation of RBT flows does not increase total biomass accrual of periphyton in LCR.

HO_{2Ceco}: Continued fluctuations of flow during the fall do not increase total biomass accrual of periphyton in LCR.

HO_{3eco}: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not increase the availability of fish food, organisms in LCR

HO_{3Aeco}: Continued implementation of MWF flows does not increase availability of fish food organisms in LCR.

$HO3_{Beco}$: Continued implementation of RBT flows does not increase availability of fish food organisms in LCR.

$HO3_{Ceco}$: Continued fluctuations of flows during the fall do not increase availability of fish food organisms in LCR.

2.0 METHODS

2.1 Study Area and Sampling Locations

The study area is located in southeast British Columbia on LCR between HLK Dam and the BBK gauging station (Figure 2-1). Kootenay River is a major tributary to LCR, and there are several smaller tributaries including Norns, Blueberry, China and Champion Creeks. The study area is divided into three reaches: 1) from HLK Dam to Norns Creek; 2) from Norns Creek confluence to the Kootenay River, and 3) from the Kootenay River confluence to BBK gauging station.

There are two types of monitoring stations, water quality index stations (WQIS) and benthic productivity sampling stations. Physical parameters including water temperature and water level were collected at six WQIS distributed within the three reaches of LCR and in the Kootenay River (Figure 2-1 and Table 2-1). Water quality sampling was not conducted in 2016, and will also not be carried out over the remaining three years of the contract, to free up budget for completion of three sessions of productivity sampling in both 2016 and 2018. Periphyton and macroinvertebrate productivity monitoring took place along depth transects (5 depths) at each of seven different productivity monitoring sites within Reach 2, during the winter, summer and fall of 2016. There were a total of 105 periphyton and benthic invertebrate samplers deployed. However, 96 periphyton samples were retrieved, whereas 90 benthic invertebrate samplers were retrieved (Figure 2-2; Table 2-2).

Table 2-1: Monitoring Stations, Sample Types and UTM Coordinates Zone UTM 11.

Station Name & General Location	Station Characteristics	Sample Type	UTM Coordinates	
			Northing	Easting
WQIS1 (across from Zellstoff Celgar Ltd.)	Upstream of Celgar outfall	Physical/chemical/water level	5,465,742	445,693
WQIS2 (upstream of boat launch)	Downstream of Celgar outfall	Physical/chemical/water level	5,464,573	450,072
WQIS3 (downstream of railway bridge)	Within back channel area	Physical/chemical/water level	5,464,517	452,244
WQIS4 (~7 km downstream of Kootenay River confluence)	Left bank off of bedrock face	Physical/chemical/water level	5,455,332	452,653
WQIS5 (~ 2.2 km upstream of Birchbank)	Right bank off of bedrock face	Physical/chemical/water level	5,450,221	448,514
WQ C1 (Norns Creek)	Within Pass Creek Regional Park	Physical/chemical	5,465,356	451,746
WQ C2 (Kootenay River)	Right bank, off of bedrock face	Physical/chemical/water level	5,462,911	454,114
R2-S1 (right bank, downstream of Robson Bridge)	Erosional, steep profile	Periphyton and macroinvertebrate substrates / temp / light	5,464,323	451,486
R2-S2 (left bank, downstream of railway bridge)	Erosional	Periphyton and macroinvertebrate substrates / temp / light	5,464,428	451,942
R2-S3 (left bank, below Brilliant Road)	Erosional, occasionally some deposition	Periphyton and macroinvertebrate substrates / temp / light	5,463,822	452,971
R2-S4 (right bank, upstream of Kootenay River confluence)	Erosional, occasionally some deposition	Periphyton and macroinvertebrate substrates / temp / light	5,463,186	452,592
R2-S5 (left bank, upstream of Kootenay River confluence)	Erosional, occasionally some deposition	Periphyton and macroinvertebrate substrates / temp / light	5,463,085	452,789
R2-S6 (adjacent to Waldie Island)	Depositional, macrophyte beds, municipal outfall	Periphyton and macroinvertebrate substrates / temp / light	5,464,256	452,488
R2-S7 (right bank, upstream of Kootenay River confluence)	Erosional, slower flows	Periphyton and macroinvertebrate substrates / temp / light	5,463,032	452,480

Table 2-2: Artificial Sampler Deployment and Recovery Rates in 2016.

Season	Reach	Site	Periphyton Samplers		Invertebrate Basket Samplers	
			# Deployed	# Retrieved (% Recovery)	# Deployed	# Retrieved (% Recovery)
Winter (Jan 12 - Mar 22) 10 weeks	2	Site 1 (S1)	5	5 (100)	5	5 (100)
		Site 2 (S2)	5	5 (100)	5	5 (100)
		Site 3 (S3)	5	5 (100)	5	4 (80)
		Site 4 (S4)	5	4 (80)	5	4 (80)
		Site 5 (S5)	5	4 (80)	5	4 (80)
		Site 6 (S6)	5	5 (100)	5	5 (100)
		Site 7 (S7)	5	5 (100)	5	5 (100)
Winter Totals			**35**	**33 (94)**	**35**	**32 (91)**
Summer (Jun 7 - Aug 16) 10 weeks	2	Site 1 (S1)	5	4 (80)	5	5 (100)
		Site 2 (S2)	5	4 (80)	5	4 (80)
		Site 3 (S3)	5	5 (100)	5	5 (100)
		Site 4 (S4)	5	4 (80)	5	5 (100)
		Site 5 (S5)	5	4 (80)	5	4 (80)
		Site 6 (S6)	5	5 (100)	5	5 (100)
		Site 7 (S7)	5	5 (100)	5	4 (80)
Summer Totals			**35**	**31 (89)**	**35**	**32 (91)**
Fall (Aug 17 - Oct 25) 10 weeks	2	Site 1 (S1)	5	5 (100)	5	3 (60)
		Site 2 (S2)	5	5 (100)	5	3 (60)
		Site 3 (S3)	5	5 (100)	5	3 (60)
		Site 4 (S4)	5	5 (100)	5	5 (100)
		Site 5 (S5)	5	5 (100)	5	4 (80)
		Site 6 (S6)	5	5 (100)	5	3 (60)
		Site 7 (S7)	5	5 (100)	5	5 (100)
Fall Totals			**35**	**35 (100)**	**35**	**26 (74)**
2016 Totals			**105**	**96 (94)**	**105**	**90 (86)**

NOTE: Lower sampler recovery rates were mostly caused by desiccation of shallow samplers when they were exposed by declining water levels.

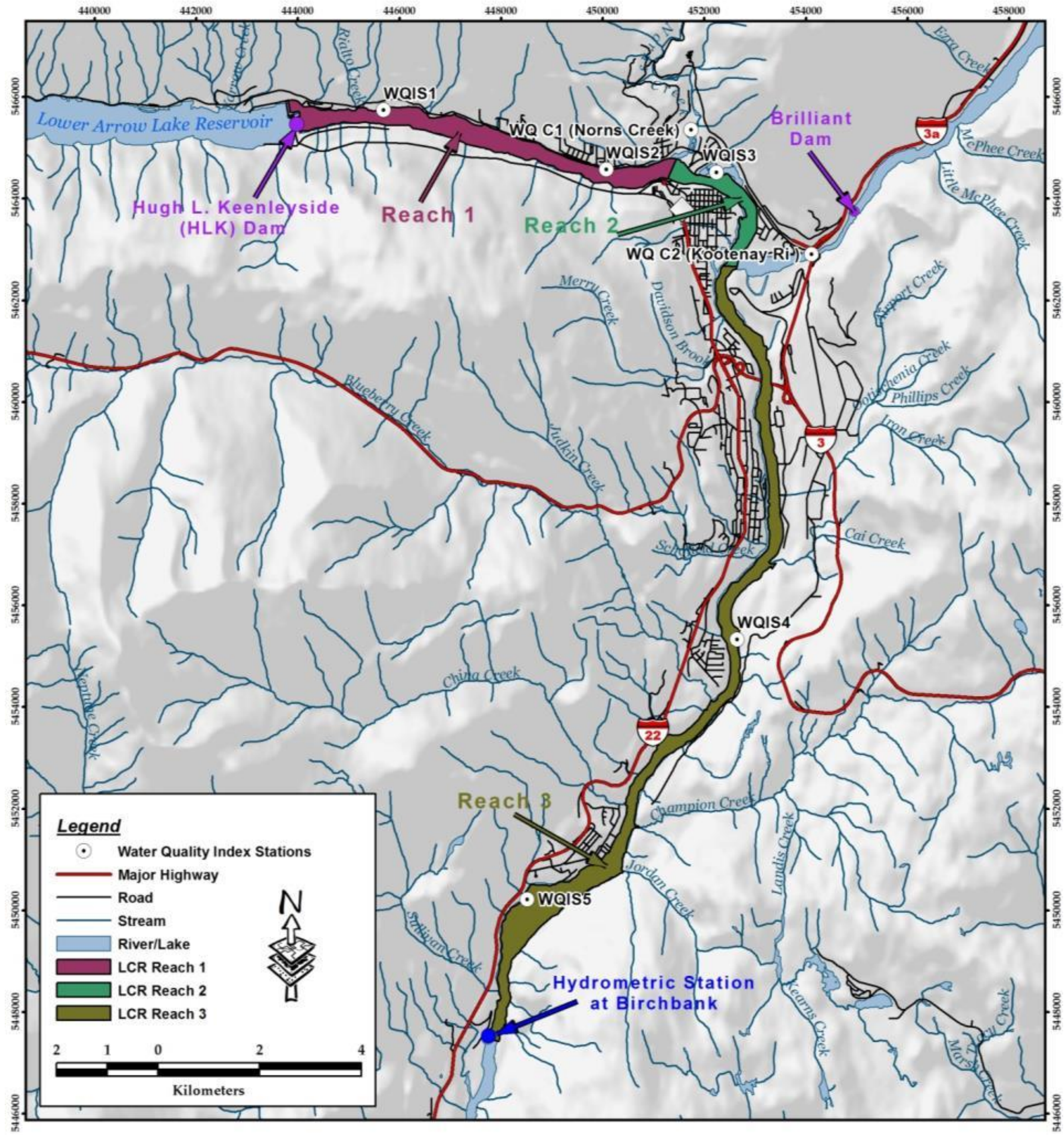

Figure 2-1: Map of Lower Columbia River Study Area and Water Quality Index Station Sampling Locations

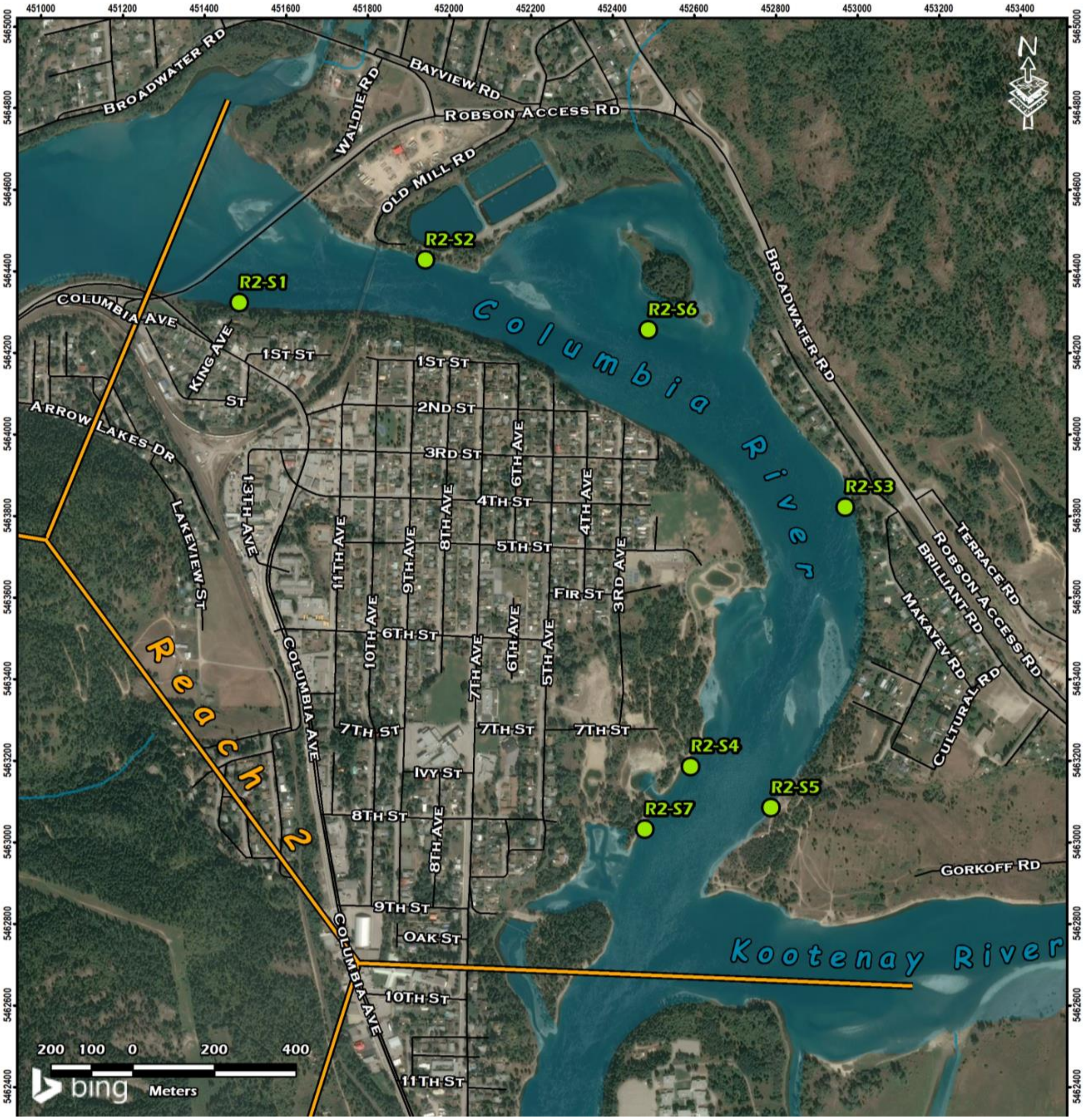

Figure 2-2: **Benthic Productivity Sampling Locations in 2016.**

2.2 Hydrology and Water Level

The Columbia River below the HLK Dam consists of flows originating from HLK Dam and the Arrow Lakes Generating Station, both of which are managed by BC Hydro. The confluence of the Kootenay tributary is located approximately 10 km downstream of HLK Dam and consists of the combined discharge (BRD) from the Brilliant Dam, the spill from Brilliant Dam, and the Brilliant Dam expansion project; each of which are managed by Fortis BC on behalf of the Columbia Power Corporation. River flows at BBK include water originating from HLK Dam, BRD Dam and all other upstream tributaries. To address the physical monitoring management question #2, river flow and discharge data were obtained from Robyn Irvine of Poisson Consulting Ltd. for all of 2016, and specific comparisons of the three different flow periods were undertaken.

Water level and temperature data were collected at five water quality index stations (WQIS1-5) within the main LCR channel, and at one station on Kootenay River (WQ C2) (Table 2-1). As previously reported, on July 19, 2011, AquiStar® PT2X Smart Sensors were installed at five WQIS1 through 5 on LCR and at one station on Kootenay River (WQ C2) (Figure 2-1). Each sensor was placed in a 1.5-inch PVC pipe that was semi-permanently mounted to either a log piling or bedrock. The AquiStar® PT2X Smart Sensors consisted of a combination pressure/temperature sensor and data logger that records data on 15 minute intervals. These sensors remained in place until the summer of 2012, when record high flows inundated the data logger component of the sensors and disabled them[1]. Replacement Onset® Water Level Logger (Model U20) pressure transducers were installed at each of the stations, except Kootenay River (WQ C2)[2], during the week of August 15 -18, 2012. The Onset logger records water levels every 20 minutes, but also requires a barologger (Model U20) to compensate for changes in barometric pressure and to measure air temperature. One barologger was installed at the top end of LCR in Reach 1 and another was installed adjacent to WQIS4 within the upland forest canopy. All pressure readings were compensated for barometric pressure and converted to water depth using HOBOware® software. Water depth was converted to elevation based on the length of the sensor cable and the surveyed elevation of the top of the stilling well.

The elevation survey of each stilling well was completed by Robert Wagner of Ecoscape Environmental Consultants Ltd. on September 21, 2011. The obtained survey data allowed for the direct comparison of sensor locations with LCR elevations.

In June 2016, it was discovered that one of the installed sensors (WQIS3) had failed and was no longer collecting accurate elevation data. Due to inconsistencies, Jan – Jun 2016 data was excluded from the master dataset. The sensor was removed and sent back to the manufacturer for repair. The repaired sensor was replaced in August 2016.

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[1] The data logger component of the sensors were positioned approximately 0.5 - 1 vertical metre above the previously documented high water level. The inundated data loggers were sent to the manufacturer in hopes of recovering lost data, but unfortunately data could not be retrieved and the units were no longer viable.

[2] The replacement sensor at the Kootenay River site could not be installed due to a continuation of high flows. The sensor was successfully mounted on September 13, 2012.

2.3 Physical and Chemical Characteristics

Historically, chemical and physical water quality parameters were collected at seven sampling locations (Table 2-1). The number of water quality sampling locations was reduced from ten to seven, as per a recommendation put forth in Year 4 (2011) when flows in Blueberry, China and Champion Creeks were recorded as minimal to nil throughout several of the sampling sessions (Olson-Russello *et al.* 2012). After 2015, it was decided to discontinue the water quality sampling program in order to free up budget for the completion of three productivity sampling sessions during 2016 and 2018.

Three LCR WQIS are located upstream of the Kootenay River confluence (WQIS1 through 3), and two below (WQIS4 and 5). Three of the five WQIS occur in proximity to noteworthy nutrient sources. WQISI occurs close to Zellstoff Celgar Mill (Celgar), a pulp processing facility, and WQIS3 and WQIS5 are located close to City of Castlegar outfalls. The City of Castlegar has two separate secondary sewage treatment systems, both authorized under Waste Management Act permits. One of the treatment systems discharges effluent into the Columbia River from the north bank, about 1 km upstream of the Kootenay-LCR confluence. The other system discharges near the west bank, 2 km downstream from the Kootenay-LCR confluence. Available effluent data indicates that discharge levels have remained below permitted maximums (Butcher 1992).

Although water quality data was not collected in 2016, historic nutrient source data from station AR 8 of the Arrow Lakes Reservoir (ALR) has been obtained and is compared to previous water quality data in this report. Monthly averages of Total Phosphorous (T-P) and Dissolved Inorganic Nitrogen (DIN) from 2008-2014 were compared using Pearson's product moment correlation.

2.4 Benthic Productivity

Benthic productivity was determined with the use of artificial substrates placed at seven sampling sites (S1-S7) within Reach 2 during three different seasons (Figure 2-2 and Table 2-1). Each periphyton artificial substrate was mounted with a HOBO Pendant temperature/light logger that continuously collected data every ½ hour throughout each deployment session. Productivity sampling in Years 5, 7 and 9 differed from Years 1-3, in that all sampling locations were in Reach 2 and were sampled during summer, fall and winter. In addition, the depths sampled at each site were increased from three depths to five. Previously, depths were referred to as shallow [S], mid [M], or deep [D]. The five depths sampled since 2012 are referred to as shallow [S], moderately shallow [MS], mid [M], moderately deep [MD] and deep [D]. The depth strata range was consistent with Years 1 – 3 (Table 2-3).

Table 2-3: Naming Convention of Sampling Depths and Corresponding Depth Strata

Depth Label	Depth Name	Depth Strata (m)
D	Deep	>5.5
MD	Moderately deep	4 – 5.5
M	Mid	2.5 – 4
MS	Moderately shallow	1 – 2.5
S	Shallow	<1

2.4.1 Periphyton and Invertebrate Sampling using Artificial Samplers

2.4.1.1 Artificial Sampler Design and Deployment

In 2016, a single artificial sampler apparatus design was used for all seasons over a 10-week sampling duration (Figure 2-3). The winter samplers were deployed from January 12th through March 22nd. The sampling session was designed to coincide with the MWF flow period. The summer sampling period occurred from June 7th through August 16th and the fall sampling period occurred from August 17th through October 25th. The winter and fall sampling sessions entirely overlap with MWF and FFF flows, while only the first month of the summer deployment overlaps with the RBT flow period. Table 2-2 provides deployment dates, sampling numbers and equipment recovery rates.

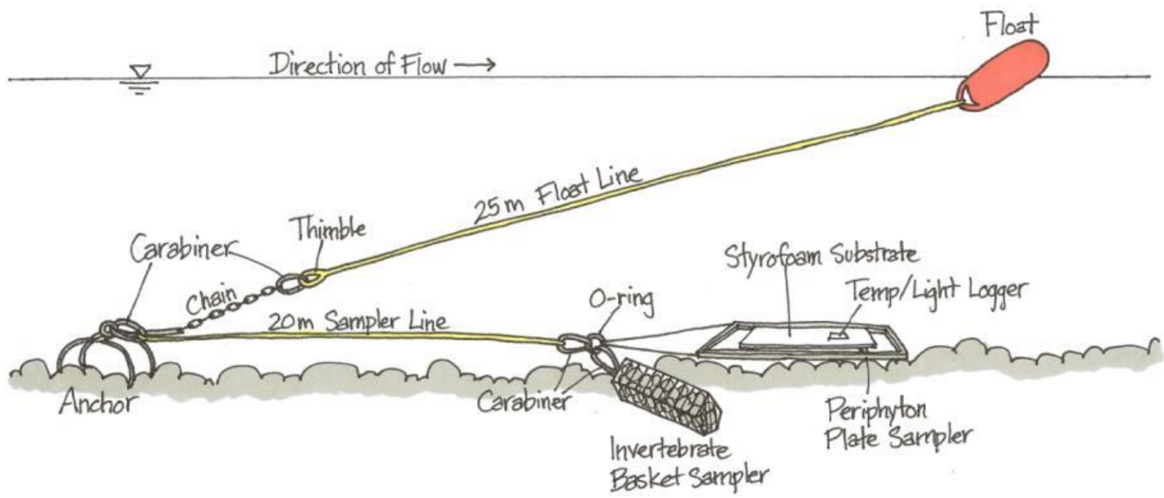

Figure 2-3: Diagram of the Periphyton and Macroinvertebrate Sampling Apparatus Deployed in Winter, Summer and Fall (2016)

To ensure the samplers were deployed right side up, a chandelier method of deployment was used (Figure 2-4). Two ropes were fastened to the corners of the steel frame so that the periphyton sampler drifted through the water column horizontally. Once positioned on the bottom, the longest rope was pulled through the apparatus and back into the boat.

Figure 2-4: The Chandelier Deployment Method

2.4.1.2 Winter Accrual Data Collection

2016 was the second year that accrual sampling, designed to investigate periphyton biomass accrual rates and test management hypothesis Ho_{2eco}, was completed during the winter sampling period. However, unlike 2014 only MS, M, and MD were included in accrual sampling. Each deployed sampler was retrieved from the river at 2, 4, 6, 8 and 10 weeks after deployment. A single periphyton punch was randomly collected from the Styrofoam and was immediately packed on ice and placed in the dark until they could be delivered to Caro Labs Kelowna for chl-a analysis. The samplers were then carefully returned to the river bottom, ensuring that the Styrofoam sat upright.

2.4.1.3 Artificial Sampler Retrieval

After 10 weeks of deployment, four periphyton Styrofoam punches were randomly collected from each sampler to assess the following metrics: 1) chl-a to give an estimate of only live autotrophic biomass; 2) Ash-Free Dry Weight (volatile solids) /total dry weight to give an estimate of the carbon component (Stockner and Armstrong 1971); and 3) taxa and biovolume to give an accurate estimate of live and dead standing crop (Wetzel and Likens, 1991). Styrofoam punches were placed in pre-labeled containers and stored on ice until further processing.

Benthic invertebrate baskets were retrieved following a similar protocol to the one described in Perrin and Chapman (2010). A 250 µm mesh net was placed beneath baskets while still in the water column to collect any invertebrates that could have been lost as baskets were lifted from the water. The net was inverted and any contents were rinsed into a labeled bucket with pre-filtered river water. The retrieved baskets were also placed in the labeled buckets until further field processing.

Upon completion of sampler retrievals from each site, individual rocks from each basket were scrubbed with a soft brush to release clinging invertebrates. Washed rocks were then rinsed in the sample water, prior to being placed back in the basket and stored for re-use. The contents from each bucket were then captured on a 250µm sieve, placed in pre-labeled containers and then fixed in an 80% ethanol solution. Detailed protocols on the retrieval and field processing of samples are available upon request.

2.4.2 Periphyton and Invertebrate Post Processing

2.4.2.1 Periphyton Post Processing

Of the four Styrofoam punches obtained from each artificial substrate, one was frozen and transported to Caro Laboratories in Kelowna, BC for the processing of low-detection limit fluorometric chl-a analysis. Another punch was chilled and transferred to Caro Labs in Kelowna, BC for analysis of dry weight and ash free dry weight (AFDW). The remaining two punches were used for taxonomic identification. Fresh, chilled punches were examined within 48-hours for protozoa and other microflora that cannot be reliably identified from preserved samples. Heather Larratt tested Lugol's solution compared to freezing the Styrofoam and determined that freezing provided enhanced long-term viability. One of the two punches was therefore frozen and stored until taxonomic identification and biovolume measurements could be undertaken. Species cell density and total biovolume were recorded for each sample. A photographic archive was compiled from LCR samples. Detailed protocols on periphyton laboratory processing are available from Larratt Aquatic.

Periphyton datasets from 2016 and previous years of the study (2008 – 2010, 2012, 2014) were standardized for statistical analyses. Eleven rare and questionable taxa were removed from the first three years of the study based on the following criteria:

1. Species not present on Dr. John Stocker's LCR periphyton taxonomy list
2. Classifications where taxonomy was questionable
3. Comprised less than 0.5% of total community in any given year
4. Comprised less than 1% of total community within any given sampler

2.4.2.2 Benthic Invertebrate Post Processing

Following retrieval, fixed benthic invertebrate samples were transported to Cordillera Consulting in Summerland BC. Samples were sorted and identified to the genus-species level where possible. Benthic invertebrate identification and biomass calculations followed standard procedures. Briefly, field samples had organic portions removed and rough estimates of invertebrate density were calculated to determine if sub-sampling was required. After samples were sorted, all macroinvertebrates were identified to species and all micro portions were identified following the Standard Taxonomic Effort lists compiled by the Xerces Society for Invertebrate Conservation for the Pacific Northwest. A reference sample was kept for each unique taxon found. A sampling efficiency of 95% was used for benthic invertebrate identification and was determined through independent sampling. Numerous keys were referenced in the identification of benthic invertebrate taxa and a partial list of references is provided in Schleppe *at al*. (2012). Species abundance and biomass were determined for each sample. Biomass estimates were completed using standard regression from Benke (1999) for invertebrates and Smock (1980) for Oligochaetes. If samples were large, subsamples were processed following similar methods. Detailed protocols on invertebrate laboratory processing are available upon request.

2.5 Statistical Procedures

All statistical analyses and the creation of most figures were conducted in R (R Development Core Team 2015). Prior to carrying out statistical analyses, 2016 data was combined with datasets from previous years (2008-2015).

2.5.1 Water Levels

The mean 2016 water level elevations recorded at WQIS1-5 in LCR and WQ C2 in Kootenay River were compared to the combined water elevation (± SD) during all years. Subsequent analysis of the effects of water level during MWF and RBT flow periods relied on the following key assumptions:

- The channel morphology has not changed substantially since pre-MWF flows (~1984), and;
- The river stage or elevation at any given WQIS can be largely predicted by flows within LCR and that small tributaries or effluent discharges have negligible effects on river elevation.

2.5.1.1 Mountain Whitefish Flow Period

To address the sub-hypothesis HO_{2Aphy}, that states continued implementation of MWF flows does not reduce the river level difference between the maximum peak spawning flow (Jan 1 to Jan 21) and the minimum incubation flow (Jan 21 to Mar 31), the water elevation difference between the maximum elevation during spawning and minimum elevation observed during incubation at each WQIS was investigated. Because historic river elevation data was not available, predicted elevations were calculated from flow data. The methods used for the 2016 analysis were the same methods used in Olson-Russello *et al.* (2015). These methods used the whole annual dataset rather than only a subset of the flow period, in order to increase the accuracy of the predicted elevations. The predicted elevations were then subsequently subset by flow period for further use in the analysis. Candidate linear regression models of water elevation were constructed for each WQIS, containing all combinations of flows from HLK, BRD, and BBK, and their associated quadratic terms (flow values^2) as explanatory variables (Table 2-4). Quadratic terms and appropriate data transformations were considered to account for potential logarithmic or non-linear relationships between flow and elevation. Model selection via Akaike information criterion corrected for small sample sizes (AICc) was used to determine the best fit and other plausible models (Δ AICc<3). In this approach, candidate models were considered and ranked based on their AICc scores. The best fit model exhibited a trade-off between model complexity and optimal fit of regression.

The top model for each site was then used to predict water elevation for periods between pre-implementation of MWF flows (1984 to 1994), post-implementation of MWF flows (1995 to 2007), and continuation of MWF flows (2008-2016). Differences among predicted elevations during each flow period were tested using a permutation ANOVA and subsequent post-hoc analysis (Tukey's HSD) to determine groupings. The permutation ANOVA was used in lieu of traditional ANOVA or Student's t tests because it does not require the same assumptions of normality, and was preferred to non-parametric methods due to ease of interpretation of results and the ability to conduct post-hoc analyses. Finally, the data were compared to actual elevations measured during 2008 - 2016 to investigate how predicted elevations compared to field collected elevations.

Table 2-4: Flow Combinations used in Regression Modeling for Predicting Water Levels during the MWF and RBT Flow Periods

Possible Predictor Flows
HLK flow
HLK flow + HLK flow ^{2}
Brilliant flow
Brilliant flow + Brilliant flow ^{2}
Birchbank flow
Birchbank flow + Birchbank flow ^{2}

2.5.1.2 Rainbow Trout Flow Period

To address sub-hypothesis HO_{2Bphy}, that states continued implementation of RBT flows does not maintain constant water level elevations at Norns Creek fan between April 1 and June 30, we used the same analysis procedure described above for sub-hypothesis HO_{2Aphy}. To limit the analysis to the Norns Creek fan, the closest two sites, WQIS2 and WQIS3, were included. To evaluate the cumulative elevation differences over the RBT flow period, linear regressions of water elevation were constructed for each site, containing all combinations of flows from HLK, BRD, and BBK, and their associated quadratic terms as explanatory variables (Table 2-4). The same model selection process was used to determine the best fit model of all plausible models (Δ AICc<3) and subsequently predict elevation during pre-implementation of RBT flows (1984 to1991), implementation of RBT flows (1992 to 2007), and continued RBT flows (2008-2016). Differences among predicted elevations during each time period were again tested using a permutation ANOVA and subsequent post hoc analysis (Tukey's HSD) to determine groupings. Finally, the data were compared to actual elevations measured in 2008-2016 to investigate how predicted values compared to those collected in the field.

2.5.2 Water Temperature

Prior to formal analyses of the effects of environmental and physical variables on LCR water temperature, exploratory analyses and development of explanatory variables were conducted. First, autocorrelation among these explanatory variables were tested using pair-wise correlation coefficients and variance inflation factors following methods outlined by Zuur *et al.* (2009). All correlation coefficients were below 0.5, and Variance Inflation Factor (VIF) scores were also low, suggesting that autocorrelation among predictors was not a concern. This allowed all possible combinations of explanatory variables to be considered in candidate models. WQIS1 through WQIS3 occur above the confluence of the Kootenay River and only experience flows from HLK whereas, WQIS4 and WQIS5 occur downstream and are subject to flows from both HLK and BRD. To account for this, associated explanatory variables were standardized based on location. Flows, reservoir temperature, and water elevation from HLK were used for WQIS1 through WQIS3 sites while BRD /BBK flows were used for WQIS4 and WQIS5 sites.

To characterize reservoir temperature as an explanatory variable, values were weighted by associated flows using the following equation:

$$T_{Res.} = \frac{(F_{HLK} \times T_{Arrow}) + (F_{BRD} \times T_{Kootenay})}{(F_{HLK} + F_{BRD})}$$

Where F is the flow for either HLK or BRD and T is the reservoir temperature for either Arrow Reservoir or Kootenay Lake. This analysis assumed that the final river temperature depends upon the total volume of water and the temperature of the two different water sources only (i.e., there are no other influences), and that all temperature measurements have occurred in a completely mixed solution of the two water sources. This formula was used for WQIS4 and WQIS5, whereas WQIS1 through WQIS3 used just Arrow Reservoir temperatures since they are above the confluence of the Kootenay River.

Likewise, reservoir elevation was calculated using the following equation:

$$E_{Res.} = \left(\frac{F_{HLK}}{F_{BBK}} \times E_{Arrow}\right) + \left(\frac{F_{BRD}}{F_{BBK}} \times E_{Kootenay}\right)$$

Where *F* is flow from HLK, BBK, or BRD, and *E* is the water elevation. Temperature data from Kootenay Lake were only available for one to two days in each season. We created a full temperature dataset for this lake to be used in subsequent analyses by predicting daily water temperature from a Generalized Additive Model (GAM) of daily water temperature. This model incorporated both point data from Kootenay Lake and a full dataset from Arrow Reservoir, with day of year (1-365), season, and location (Kootenay Lake or Arrow Reservoir) as explanatory variables. Similar to temperature, this formula was used for WQIS4 and WQIS5, whereas WQIS1 through WQIS3 used just Arrow Reservoir elevations since they are above the confluence of the Kootenay River.

The data was separated by flow period (MWF, RBT, FFF). The following analysis was performed for each flow period. We used linear mixed-effects modeling (Zuur *et al.* 2009), model selection via AICc to evaluate the relative effects of water temperature and elevation from above site reservoirs, flow from dams (HLK and BRD), Castlegar air temperature and seasonal flow period on LCR water temperatures. In this approach, candidate linear mixed-effects models containing all combinations of the above explanatory variables were constructed with sampling site and year included as random effects to account for the potential lack of independence among measurements from the same year or site. Candidate models were then competed in AICc model selection process described above for elevation and flow period analyses. We also calculated pseudo R^2, derived from regressions of observed data versus fitted values (Cox and Snell 1989; Magee 1990; Nagelkerke 1991; and Piñeiro *et al.* 2008), as a measure of the variation in observed water temperatures explained by a given model. This approach ensured that all plausible explanations for water temperature were equally considered, to better understand the specific effects of flow period on water temperature.

2.5.3 Benthic and Periphyton Community Analysis

Non-metric multidimensional scaling (NMDS) was used to explore variation in benthic and periphyton community composition at the genus level. The Bray-Curtis dissimilarity index was used for both NMDS analyzes. This index is sensitive to the variation of species that have smaller abundances (Clarke and Warwick 1998). To visually explore differences in community compositions, the NMDS scores for every sample site from 2008-2016 were plotted using R package ggplot2 (Wickham 2009). A permutational multivariate analysis of variance (PERMANOVA) was used to determine if there were significant differences in community compositions according season. The amount of variability in community composition explained by season was determined by calculating the partial R^2 from a permutational MANOVA. Both NMDS and permutational MANOVAs do not make assumptions of the variable distributions and relationships (Anderson 2001; Clarke *et al.* 2006). The NMDS analysis and permutational MANOVA used R package vegan version 2.3-5 (Oksanen *et al.* 2016). For both periphyton and invertebrates, the NMDS analysis was performed with rare taxa included and excluded and both results were very similar. The results presented are with rare taxa included.

2.5.4 Periphyton and Benthic Invertebrate Production

Exploratory analysis of production responses to predictors was completed for raw or log-transformed data using scatterplots for all response – predictor combinations. These plots were completed for summer, fall and winter. This graphical representation of data was used to assess the quality and general patterns in relationships and gauge the applicability of potential explanatory variables prior to their inclusion in the main statistical analyses. Table 2-5 provides a description of the explanatory variables used for both periphyton and benthic invertebrates.

Table 2-5: Explanatory Variables for both Periphyton and Benthic Invertebrates

Variable	Description
Velocity	Velocity was measured on the day of deployment and the day of retrieval. The average of these two values was used in the analysis.
Substrate Score	Substrate score numerically describes the substrate size at the plate location. It is a weighted average where higher scores are representative of larger substrates.
Flow Daily SD	The mean of the standard deviation of daily flow from HLK. This predictor describes the average daily variation of flow and was used to estimate the effects of flow regulation during the FFF flow periods.
Elev. Diff (MWF)	Only calculated for the MWF flow period, it is equal to the elevation difference of maximum elevation during the spawning period (deployment-Jan 21) and minimum elevation during the incubation period (Jan 22-retrieval)
Elev. Diff (RBT)	Only calculated for the RBT flow period, the sum of elevation drops for the deployment-retrieval dates that coincide with the RBT flow period. Elevation drops are calculated by daily differences in mean elevation.

Temperature and light were originally considered. When separated by season, the fluctuation in temperature and light were minimal. This caused the light and temperature to have little effect and be overshadowed by other predictors in the model, therefore these explanatory variables were dropped from the analysis. Explanatory variables were standardized to allow for direct comparison.

The response variables for periphyton and benthic invertebrates are described in Table 2-6 and Table 2-7. Upon inspection of the residual plots for periphyton: total abundance, and total biovolume were log transformed to reduce heteroscedasticity. To further ensure that models met the assumption of normally distributed residuals, Cook's distance and residual plots were examined to identify influential samples. For most models 1-3 samples were removed because they were having a leverage effect (Appendix-A8 and A10). We acknowledge that model pseudo R^2 may be over predicted because extreme values have been removed from analyses.

Table 2-6: Responses for Periphyton

Variable	Description
Total Abundance	Total Abundance across all species
Total Biovolume	Total Biovolume across all species
Chl-a	Total Chlorophyll-a

The following datasets for benthic invertebrates were log transformed: total abundance, total biomass, percent EPT, and percent quality forage (Table 2-7).

We used linear mixed-effects modeling (Zuur *et al.* 2009) and AICc model selection to evaluate the relative effects of the predictors on each response. We used the MuMlin package in R (Barton 2012) to complete the models based on ΔAICc values and AICc weights (w_i), and to calculate multi-model averaged parameter estimates from 95% confidence sets for each response variable (Burnham and Anderson 2002; Grueber *et al.* 2011). We calculated relative variable importance (RVI), which is the sum of AICc weights from all models containing the variable of interest with variables having RVI values above 0.55 and confidence intervals that did not span zero.

This year Didymo was modelled to identify the potential drivers of Didymo abundance in the LCR. The presence of Didymo in a periphyton community causes increases in total biovolume and may result in a decrease of benthic invertebrate and periphyton abundance. Didymo modelling was conducted to test if flow variability causes differences in Didymo and therefore alters periphyton productivity. Classification and Regression Tree (CART) were used because these models accommodate multicollinear predictors and non-parametric distributions (De'ath and Fabricus, 2000; Elith *et al.* 2008). The CART model was run with the following predictors: mean flow (calculated over each flow period of interest), mean daily light intensity (light), mean daily water temperature (temp), flow daily SD, velocity, site, depth, season, and year. Relative abundance of Didymo was used as the response variable and all seasons and years were included. The CART algorithm works by partitioning the data into groups based on a split point and a splitting variable (i.e. an explanatory variable). The split point and variable is determined by searching through every possible combination of explanatory variables and values (Hastie *et al.* 2001). The split point that is selected is the one that minimize differences within nodes (i.e. groups) (De'ath and Fabricus, 2000). The CART algorithm continues to make binary splits at each tree node until a stopping criterion is reached (Elith *et al.* 2008; Jun, 2013). The stopping criterion is usually based on a cost-complexity criterion which considers the tree size and goodness of fit (Hastie *et al.* 2001). The R package partykit was used for CART modelling (Hothorn and Zeileis, 2015).

Model averaging of key periphyton responses included total abundance, biovolume, and chl-a. Data from all reaches were modelled (R1 through R3), but data from 2008 was excluded from the analysis because many of the explanatory variables were not collected in that year. For each response, the following explanatory variables were used: velocity, substrate score, and a measure of flow variability specific to the flow period of interest. A detailed description of each explanatory variable is included in the methods (Table 2-5). The number of plausible models (those with an AICc<3.0) ranged from 4 to 11 across all seasons and the total number of models considered was 96 for each season (Appendix A-8).

Table 2-7: Responses for Benthic Invertebrates

Variable	Description
Total Abundance	Total Abundance across all species
Total Biomass	Total Biomass across all species
Simpsons Index	A measure of species richness that takes into account the abundance of each species
Percent EPT	Calculated by summing the biomasses of Ephemeroptera, Trichoptera and Plecoptera, then dividing by the total biomass.
Percent Quality Forage	Calculated by summing the biomasses of Ephemeroptera, Trichoptera, Diptera, and Plecoptera, then dividing by the total biomass.

2.5.5 Fish Food

To better inform the testing of HO_{3eco}, that states operations do not increase the availability of fish food organisms in the LCR, stomach contents of RBT and MWF were analyzed. The benthic invertebrate community composition of RBT and MWF were analyzed at the family level by Cordillera Consulting in Summerland BC. The fish stomachs were from juvenile and adult RBT and MWF caught in fall of 2012 and 2014. An NMDS using the Bray-Curtis dissimilarity index was conducted on the fish stomach community data. A PERMANOVA was used to determine if there were significant differences in community compositions according to year, species, or age (mature or juvenile). To identify unique taxa in fish stomachs, taxa were related to the community differences by fitting them to the ordination plot as factors using Envfit (Oksanen *et al.* 2016). Only the taxa that were significant ($p<0.05$) and had r^2 greater than 0.1 were considered. These taxa describe the most observed variation between fish stomachs. Relative abundances of benthic invertebrate taxa were also calculated to identify dominant taxa. Dominant taxa could either be the most abundant taxa during the sampling period or the taxa that the fish prefer to consume.

As in previous years, two response variables for the benthic invertebrate models were designed to specifically test the availability of food for juvenile and adult MWF and RBT. They included % biomass of Ephemeroptera, Plecoptera and Trichoptera (EPT), and good quality forage (percent biomass of EPT + Diptera).

3.0 RESULTS

3.1 Hydrology

3.1.1 River Flows

Flow within the study area are dominated by discharges from HLK Dam on the Columbia River and from Brilliant Dam on the Kootenay River. In 2016, contributions made to the mean daily river flows from the Columbia and Kootenay Rivers were typical, at 50.9% and 45.5%, respectively, of the total flows measured at the Birchbank gauging station. The remaining 3.5% was contributed by smaller tributaries such as Norns Creek and outfalls.

Overall, the flows and water levels during the 2016 study period were low compared to previous years (2008-2015). The highest flow recorded at the Birchbank gauging station in 2016 was 3142.7 m^3/s on June 14th, which was lower and earlier than the previous five years in which higher flows peaked in July (Olson-Russello *et al.* 2015; Olson-Russello *et al.* 2014, Larratt *et al.* 2013; Olson-Russello *et al.* 2012) (Appendix-A1). This shift was weather-driven by the unusually early, hot spring weather in 2016.Figure 3-1 compares the 2016 and 2008-2016 hydrographs of mean daily river flows from LCR at HLK Dam, and the Kootenay River from the Brilliant Dam. The mean daily river flows at HLK Dam were greater than those at Brilliant Dam (989.6 m^3/s and 884.3 m^3/s, respectively).

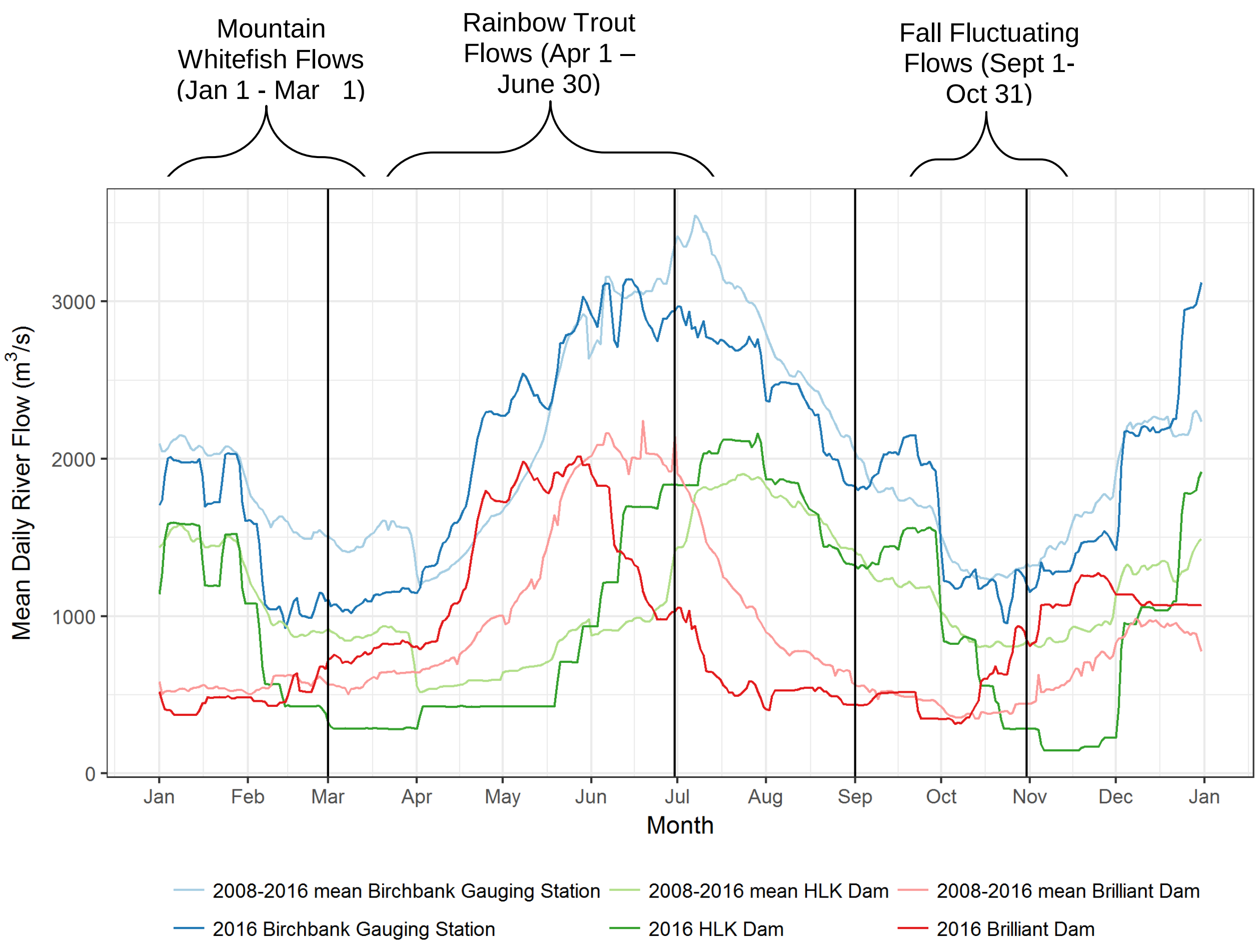

Figure 3-1: Mean daily river flow at HLK Dam (Columbia River), and Brilliant Dam (Kootenay River) in 2016 compared to the mean of 2008 – 2016 flows.

Mean daily flows were separated and summarized for MWF, RBT and FFF periods to more thoroughly understand LCR flows during each of the designated flow periods (Appendix-A1).

During most of the 2016 MWF flow period (Jan 1 – Mar 31), flows at HLK Dam, Brilliant Dam and the Birchbank gauging station exhibited a normal flow pattern. However, in March 2016, flows from HLK Dam stayed at levels that are the lowest levels seen since March 2008 when this study began. Flows from HLK Dam remained moderate (~1500 m^3/s) throughout January and began dropping on January 28th and tapered down by ~81% through the 57-day flow period to ~280 m^3/s on March 25th (Figure 3-1). Flows from HLK Dam remained below 500 m^3/s until May 20th. Flows from Brilliant Dam were similar to previous years, with consistent flows throughout the first part of the MWF flow period of ~450 m^3/s. There was a steady increase in flows from Feb 15th to March 31st, and on March 27th, flows from Brilliant Dam reached 846 m^3/s. The MWF flow period is split into spawning (Jan 1 – Jan 21) and incubation (Jan 22 – Mar 31). At the time of this writing, it was not known whether the drop in flows from HLK was large enough to expose MWF eggs.

During the 2016 RBT flow period (Apr 1 – Jun 30), flows at HLK Dam were held stable until May 20th and then increased until the end of the flow period, reaching 1846 m^3/s on June 30th. Minimum and maximum flows from HLK Dam during the 2016 RBT flow period were more extreme compared to previous years (Figure 3-1). RBT flows at Brilliant Dam steadily increased from Apr 1 through May 31 when they peaked at 2,018 m^3/s. The flows then declined over the remainder of the flow period until they reached ~980 m^3/s.

During the fall fluctuating flow period, a downward trend of mean daily flow for HLK was observed. Flows from Brilliant Dam were minimal with small fluctuations at the beginning of the flow period. Flows started to increase on October 9th and reached a maximum of 937 m^3/s on October 28th.

3.1.2 Water Levels

The following results address $Ho2_{phy}$ which investigates the influences of fish flows on river water levels.

Overall, the water levels in 2016 were below normal, particularly in late winter. Mean daily water levels at stations 1-5 were similar to previous years at the start of the MWF flow period (Figure 3-2). However, in February-April 2016, stations 1-5 had a large drop in water levels leaving sensors at WQIS1, WQ1S2, WQ1S4, and WQ1S5 exposed to air and unable to record data. This data indicates that February to April 2016 had lower mean daily water levels compared to previous years. During the RBT flow period, mean daily water levels of stations 1-5 were within the range of previous year's water levels. Like the MWF period, stations 1-3 had similar mean water levels at the beginning of the FFF period and the water levels dropped in the middle of October until the loggers were exposed. The low flow levels in October were smaller at the downstream stations 4 and 5 relative to the upstream stations (Figure 3-2).

The Kootenay station (WQ_C2) had typical water levels during the MWF period, followed by much higher water levels than previous years during the RBT flow period. The mean daily water level of WQ_C2 during the FFF period were similar to previous years (Figure 3-2). In 2016, successfully recorded water level elevations above the Kootenay River confluence ranged from ~417.5 to 421.9 m asl. Below the confluence (WQIS4 and 5), elevations ranged from ~410.7 to 416.5 m asl. The maximum mean daily river flows recorded in 2016 were 3142.7 m^3/s on June 14th. For comparison, flows recorded in previous years of this study (2011, 2012, 2013 and 2014) were 4,155.4 m^3/s on July 9th 2011; 6,043.1 m^3/s on July 21th 2012; 4,434.4 m^3/s on July 5th 2013; 3,677.9 m^3/s on July 8th 2014.

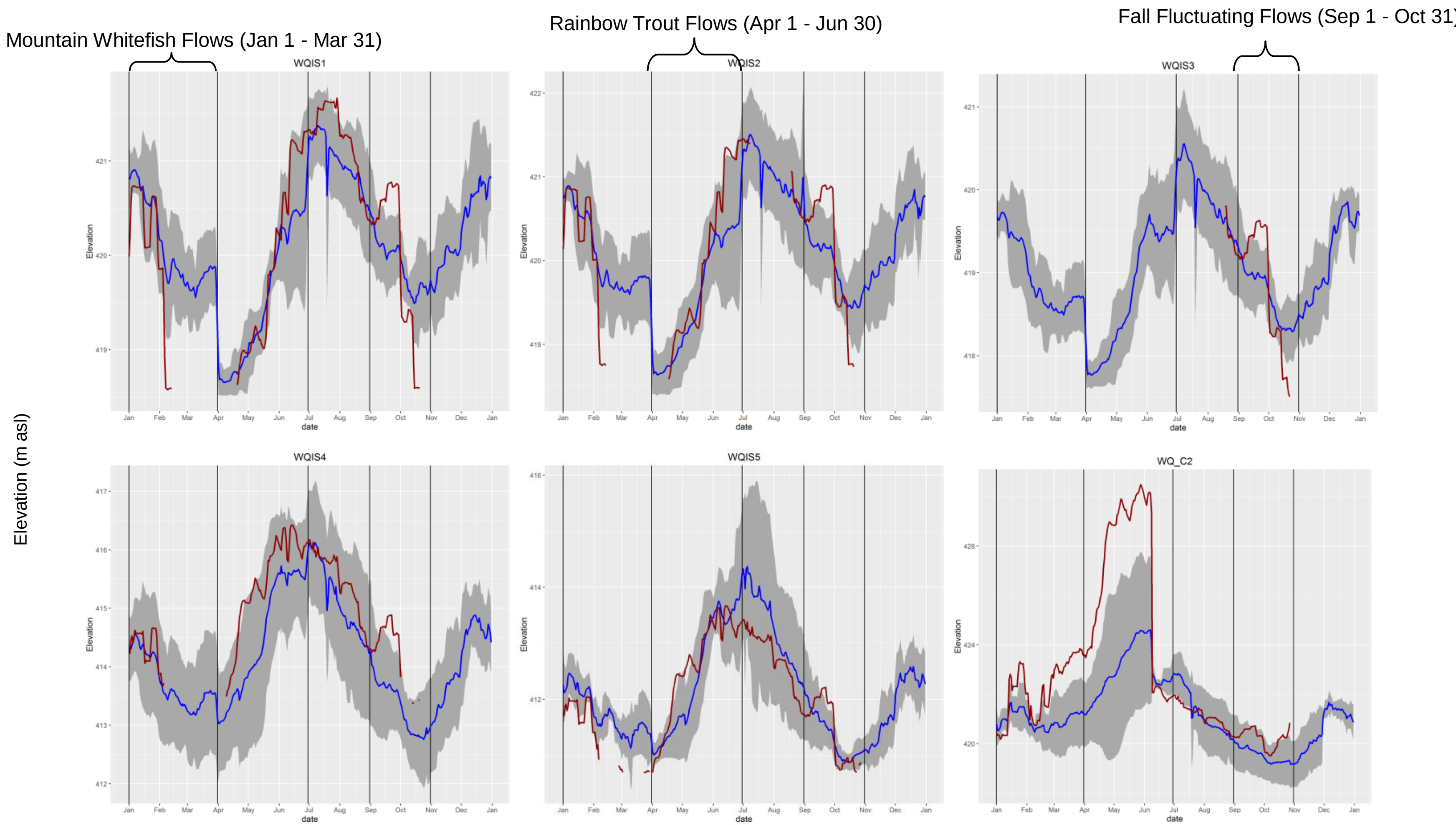

Figure 3-2: **Mean daily water levels recorded at WQIS1 – 5 on LCR and at WQ C2 on Kootenay River. The red line depicts the mean daily water level recorded at each site in 2016.** The blue line is the mean daily water level throughout the duration of the study (2008-2016± SD (gray shaded area). The SD is shown to highlight the variation in the data over multiple years, but it could not be determined for all months due to gaps in data collection.

3.1.2.1 Mountain Whitefish Flow Period (Ho_{2Aphy})

The following results address Ho_{2Aphy} which investigates the influence of MWF fish flows on river water levels.

All relationships between flow and river water levels were statistically significant ($p < 0.05$). At all the WQ sites, the predicted elevation difference during pre-MWF flows (1984-1994) was significantly higher than the predicted elevation difference during post and continuous flow periods (permutation ANOVA, d.f. 3, $p<0.001$). The accuracy of the predictive elevations is supported statistically and by comparing field measured elevations to the predicted elevations during the post-implementation period (Figure 3-3).

Statistical analyses of the flow and water elevation data indicate that the implementation of MWF flows has been effective at reducing the difference between maximum flow during MWF spawning and minimum flow during MWF incubation and should benefit the fishery (Appendix-A2). These results are consistent with findings by Scofield *et al.* (2011).

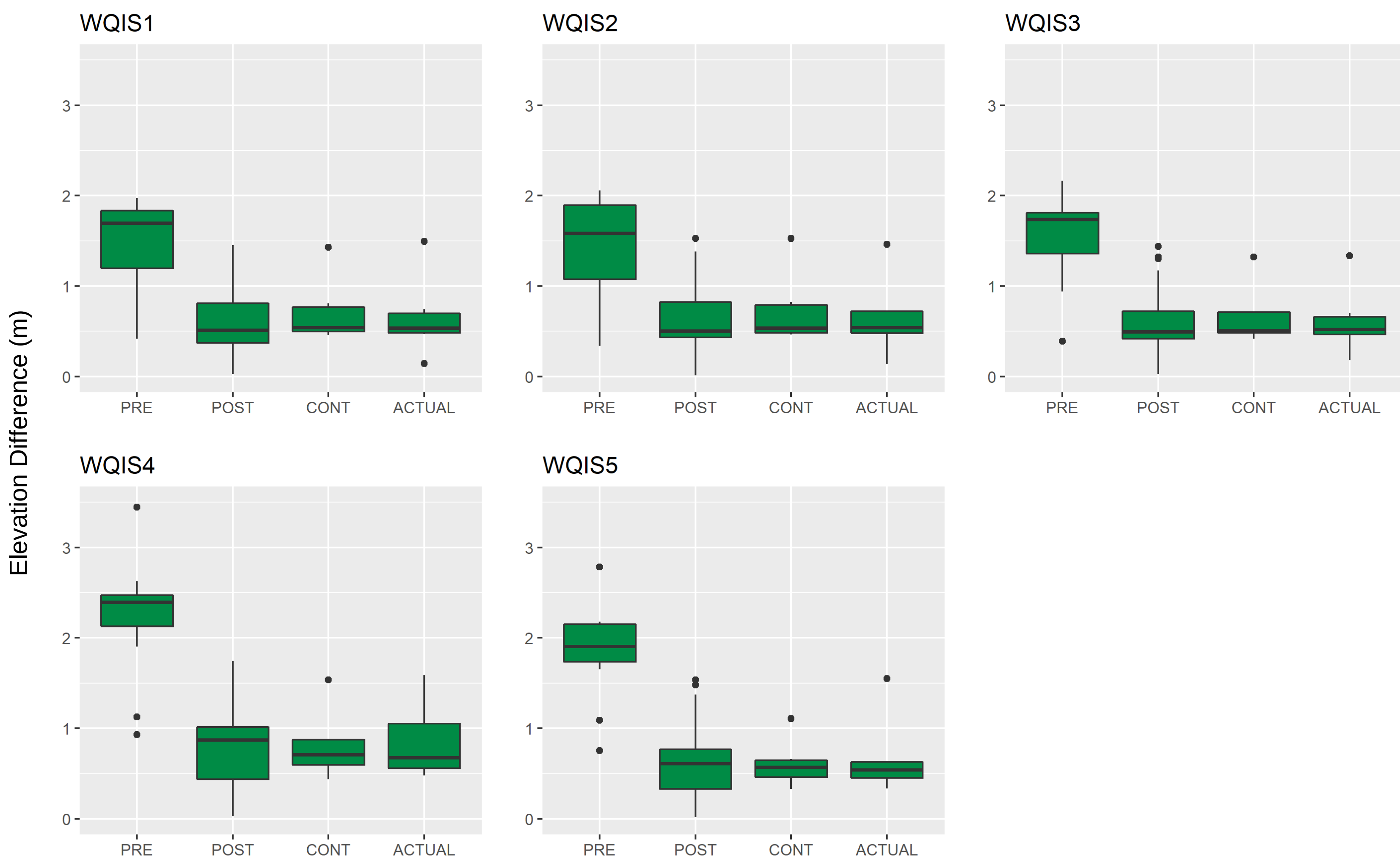

Figure 3-3: **Predicted water level elevation difference between maximum flows during Mountain Whitefish (MWF) spawning (Jan 1 – Jan 21) and minimum flows during MWF egg incubation (Jan 22 – Mar 31) for Pre (1984 – 1994), Post (1995-2007), and Continuous (2008-2016) flow years at each water quality index station.** The actual dataset is included to illustrate variability between the predicted continuous (CONT) values and actual elevation field data collected during the 2008-2016 study period.

3.1.2.2 Rainbow Trout Flow Period (Ho_{2Bphy})

The following results address Ho_{2Bphy}, which investigates the effects of RBT flows on water levels at Norn's Creek Fan between April 1 and June 30. The results are derived from analyses described in the previous section. The best statistical models for the sites WQIS2 and WQIS3 that are located near Norn's Creek Fan included BBK, BRD and HLK flowsError! Reference source not found.. At both sites, flow had a strong positive effect on ater elevation.

For both WQIS, the total elevation drop that occurred was significantly higher during pre-implementation of RBT flows (1984-1991) than during post (1992-2007) and continuous (2008-2016) flow periods (permutation ANOVA, d.f. 3, $p<0.001$, Figure 3-4). Like the results for MWF, RBT data shows good agreement between predicted and observed water elevations.

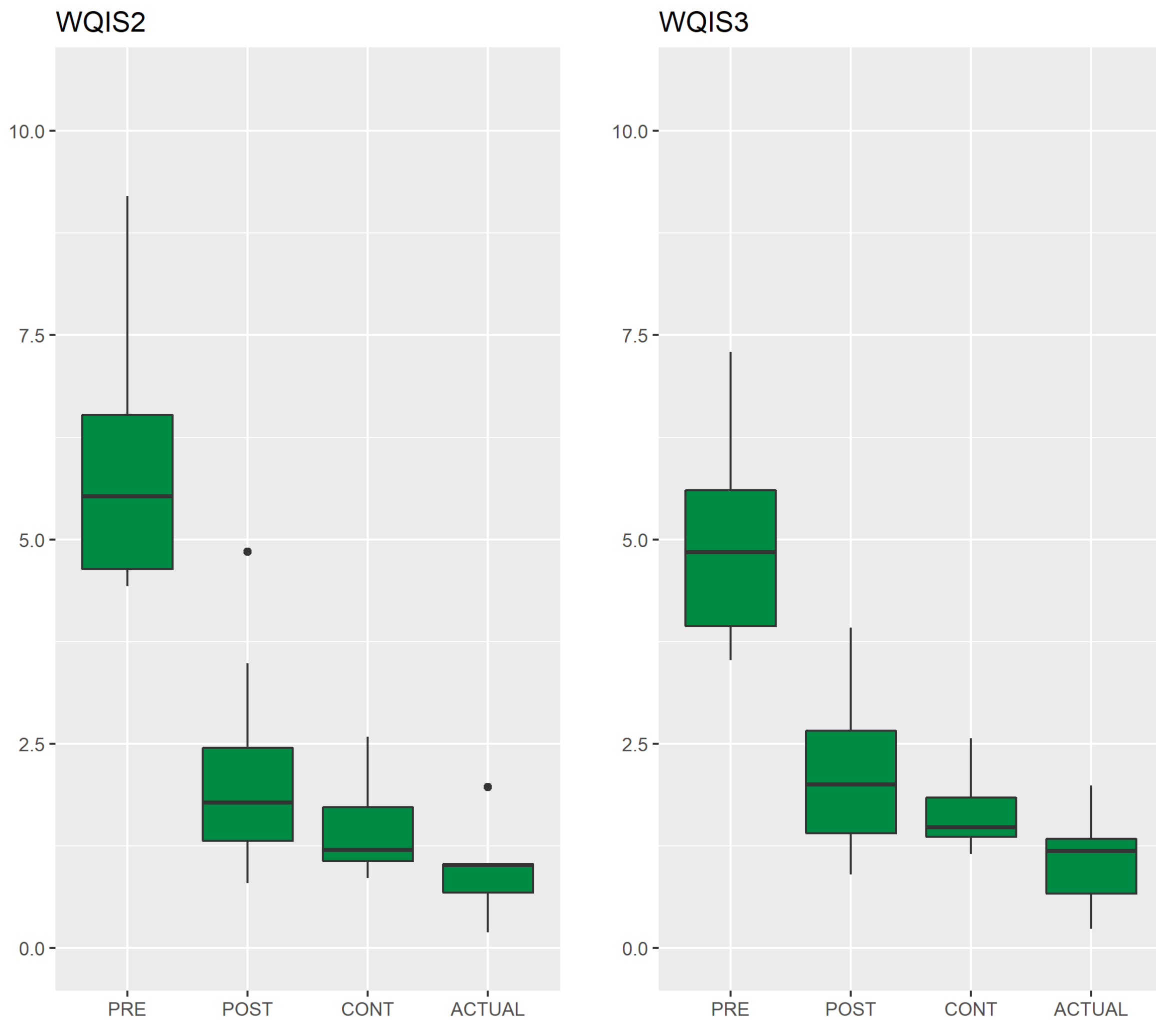

Figure 3-4: Cumulative sum of elevation drops occurring during the Rainbow Trout Flow period for Pre (1984 – 1991), Post (1992-2007), and Continuous (2008-2016) flow years at each water quality index station. The actual dataset is included to

illustrate variability between predicted CONT values and actual elevation field data collected during 2008 -2016.

3.2 Physical and Chemical Characteristics

3.2.1 Water Temperature

As with the water elevation data, 2016 water temperature data also had data gaps for sites 1-5 due to exposed sensors (Figure 3-5). Water temperatures at the five LCR WQ stations varied seasonally, and ranged from approximately 3.1 to 19.5°C during 2016. Temperatures in Kootenay River (WQ C2) were slightly higher, and ranged from approximately 3.5 to 20.5°C.

The 2016 summer daily water temperatures were similar to the mean temperatures recorded during previous years of this study. WQ stations 4 and 5 exhibited a higher variability than sites WQIS1 - 3, likely due to the influx of flows from Kootenay River. Olson-Russello *et al.* (2012), Larratt *et al.* (2013) and Olson-Russello *et al.* (2014) reported slightly higher water temperatures originating from Kootenay River compared to LCR, and it appears that the higher temperatures are responsible for increased variability in temperature observed at downstream sites.

Water temperatures follow a seasonal pattern. At the beginning of MWF flows (Jan 1 – Mar 31), the 2016 water temperatures had very little variation and were similar to previous years, between 4 and 5°C. Temperatures during the RBT flow period (Apr 1 – Jun 30) steadily increased from approximately 6 to 16°C. At stations 1-3 there were higher temperatures at the beginning of the RBT flow period compared to previous years. Finally, the FFF period exhibited the opposite trend with water temperatures declining from approximately 18 to 10°C, as they do each year.

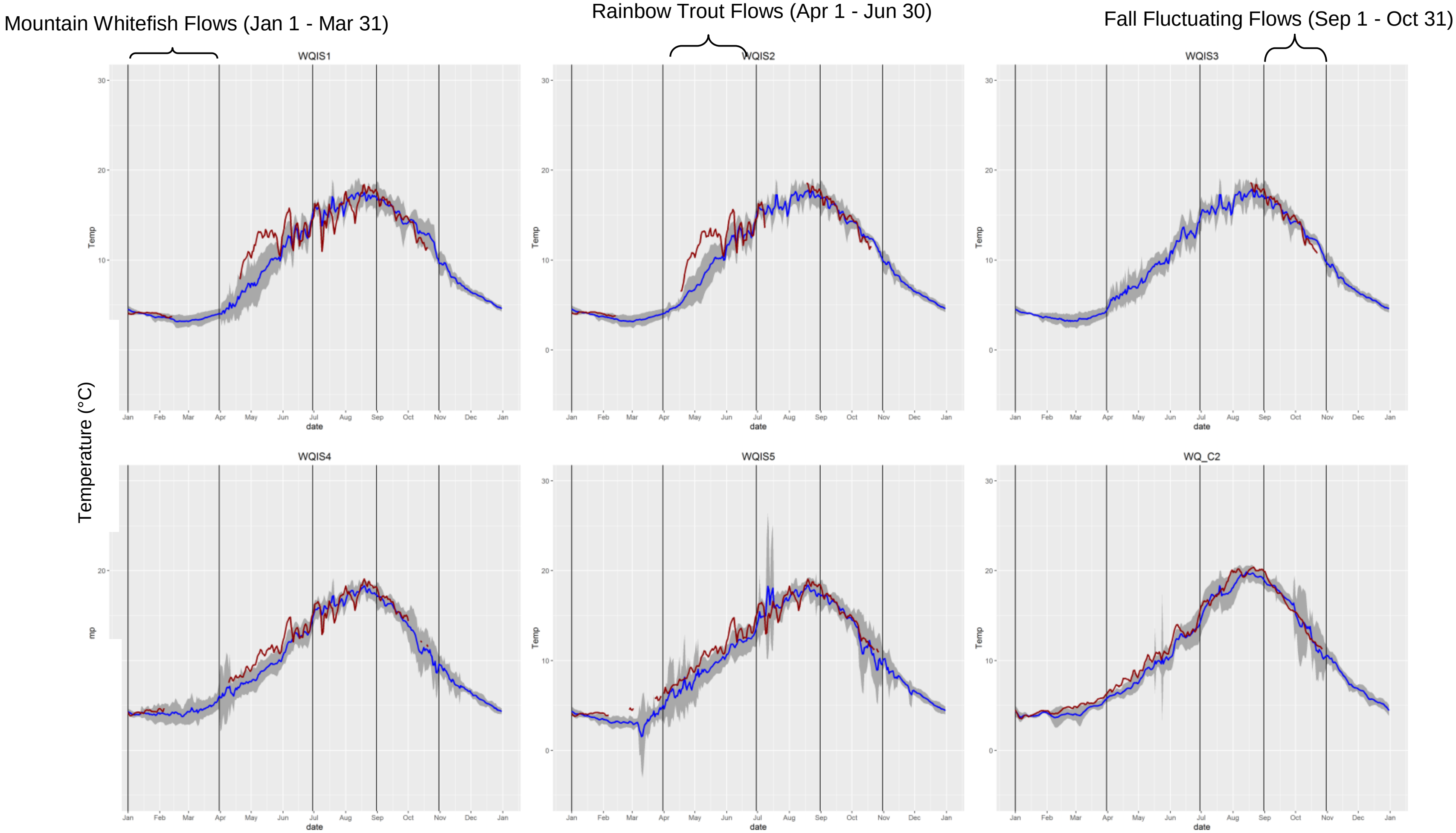

Figure 3-5: **Mean daily water temperatures recorded at WQIS1 – 5 on LCR and at WQ C2 on Kootenay River. The red line depicts the mean daily water temperature recorded at each site in 2016.** The blue line is the mean daily water temperature throughout the duration of the study (2008-16) ± SD (gray shaded area). The vertical lines indicate the beginning and end of each flow period.

To test the specific hypotheses that implementation of different flow periods may affect water temperature, we ranked the relative importance of flow regime with other parameters that may affect water temperature using statistical tests. LCR water temperatures were most strongly correlated with Castlegar air temperature during the FFF flow period and with source water reservoir temperatures during the MWF and RBT flow period (Figure 3-6 and Appendix-A3).

Reservoir water levels also affected LCR water temperature, particularly during the winter and summer, when LCR temperatures increased with increased reservoir elevation. The effect of flow on water temperature was also evident, and negatively associated during all flow periods. Based on this analysis, flow is not the most important determinant of river temperature. Reservoir temperature in the winter and summer and air temperature in the fall were stronger predictors of LCR water temperature (Figure 3-6 and Appendix-A3).

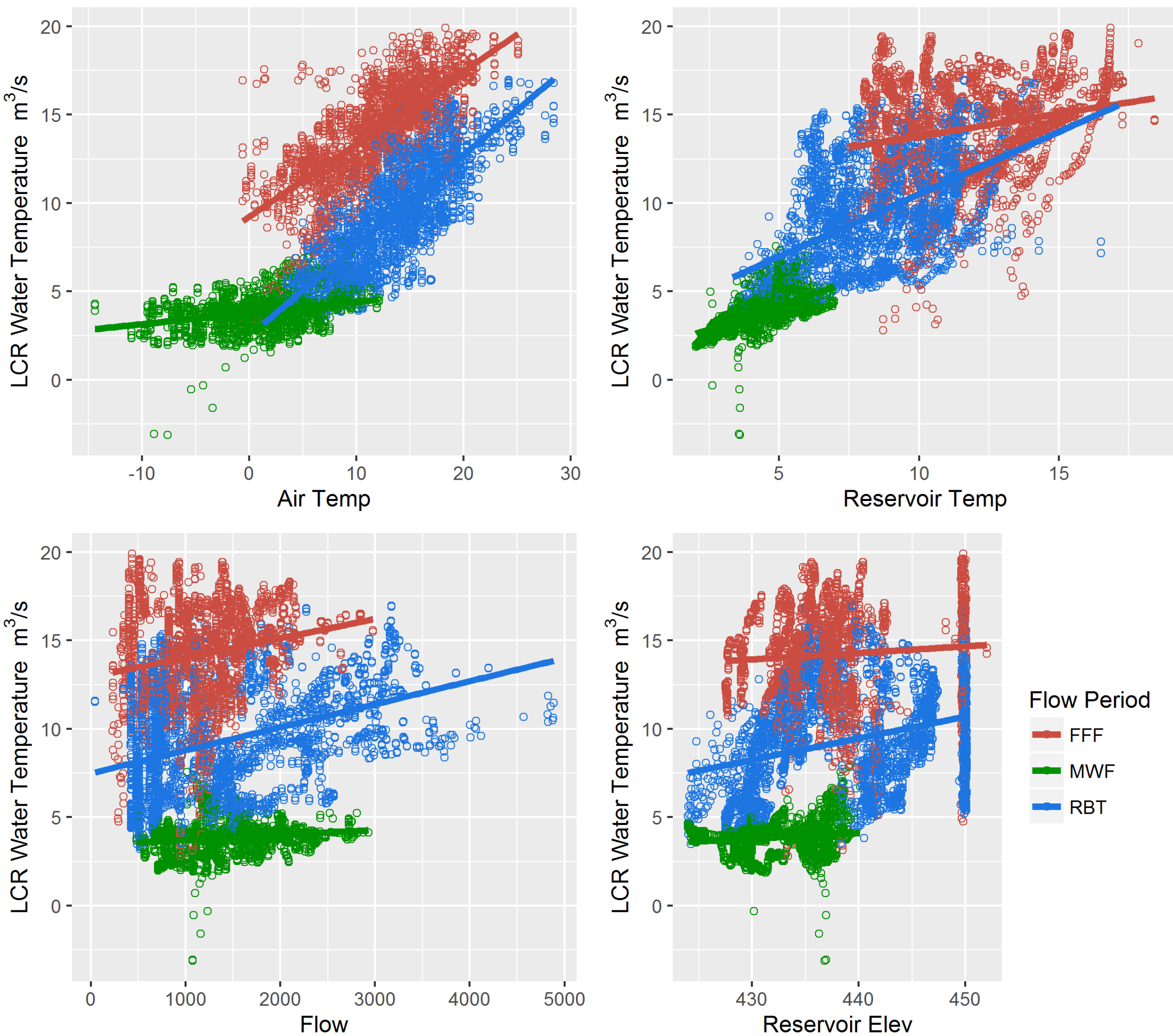

Figure 3-6: **Single linear regressions of air temperature, reservoir elevation, reservoir temperature and flow on LCR water temperature in each flow period for 2008-2016.** Flow periods are defined as FFF = Fall fluctuating flows (red), MWF = Mountain Whitefish flows (green), RBT Rainbow Trout flows (blue).

3.2.2 Water Quality

Water quality sampling was undertaken from 2008 to 2014 to address Physical Habitat Monitoring Management Question 3. The sampling was since suspended because the sampling program was not robust enough to provide statistically valid results and fiscal efforts were deemed to be more beneficial elsewhere. With the data collected to date, we tentatively accept the management hypotheses $HO_{3phy,}$ $HO_{3Aphy,}$ and HO_{3Bphy} and assume that MWF, RBT or FFF flows have no detectable effect on the water quality of LCR (Olson-Russello et al. 2015). As part of this year's analysis, potential effects of nutrient enrichment at the closest ALR station (AR-8) upon LCR nutrients was investigated. The effect of City of Castlegar municipal effluents on downstream sites could not be investigated because of insufficient nutrient data.

Average monthly Dissolved Inorganic Nitrogen (DIN) at WQIS1 was positively correlated with average monthly DIN nutrient measurements at AR-8, (r=0.56, p=0.005 (Appendix A-4)). The correlation between average monthly T-P at WQIS1 and T-P nutrient additions at AR-8 was not significant (r=-0.16, p=0.47).

3.3 Periphyton

Periphyton sampling is focused on the most productive area of the river - the permanently wetted, shallow substrates in LCR Reach 2, from the water's edge to depths of 5 - 6 m. The samplers were distributed as widely as possible at each site but none could be deployed in the deepest thalweg areas that frequently exceeded 10 m depth. Overall, periphyton growth in this key production area would classify LCR as moderately productive. Species diversity and the Simpson's index indicate that LCR biodiversity is stable and moderate compared to other large rivers (

Table 4-1).

3.3.1 Periphyton Accrual

Although periphyton metrics were variable between seasons and years, the chl-a data continues to suggest that accrual reaches peak biomass in 6-7 weeks in summer, greater than 8 weeks in fall and greater than 10 weeks in winter (Figure 3-7).

When samplers were deployed for longer than these periods, a combination of sloughing due to flow change, grazing, and shading by surface algae layers and bacterial decomposition of algae cells deep in the periphyton biofilm, all act to limit the standing crop of periphyton in LCR. Mid-depth samplers were deployed in winter 2013 for 12 and 26 weeks and chl-a peaked at 12 weeks, although periphyton biovolume continued to climb (Appendix-A5). In the 2014 winter deployment, both chl-a and biovolume were lower at 20 weeks than at 10 weeks.

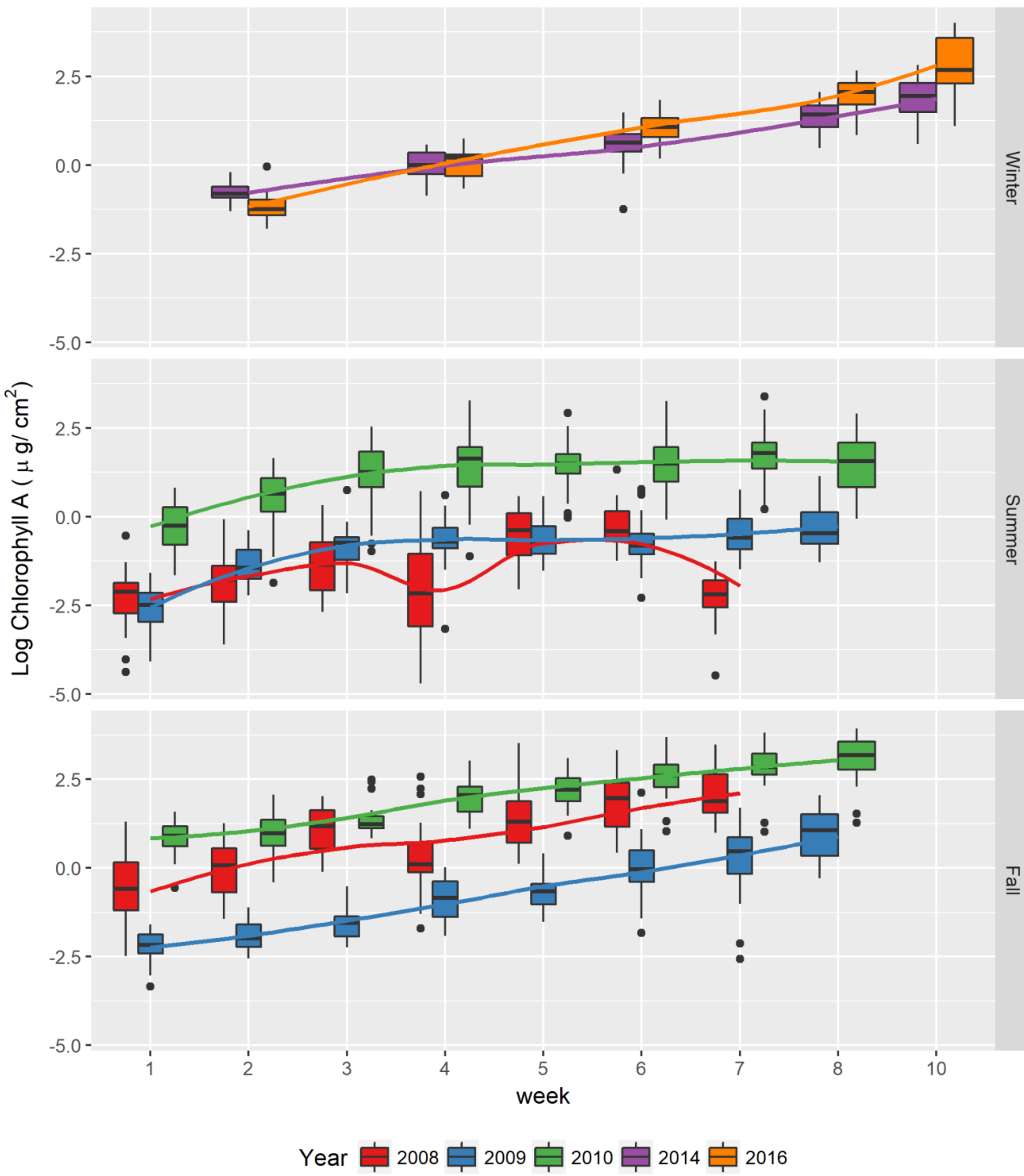

Figure 3-7: **Weekly periphyton chl-a accrual rates in the summer (2008 – 2010), the fall (2008 – 2010) and winter (2014, 2016).** Fitted lines were generated using a locally weighted polynomial regression method (LOWESS). The first three years of data were obtained from Scofield *et al.* 2011.

3.3.2 Summary of Periphyton Community Composition and Productivity in the LCR

A total of 75 periphyton algae taxa were frequently observed in LCR, while the remaining 46 observed taxa occurred at less than 5% of the sites. Like most large rivers, LCR periphyton was dominated by diatoms representing between 55 and 99% of the average biovolume in all sample sites and seasons (Appendix-A6)Error! Reference source not ound.. Over the years of study, the largest shifts in community structure occurred in the soft-bodied algae. For example, flagellate abundance oscillated over the sample periods and ranged from 0 – 43%. Filamentous cyanobacteria ranged from 0.6 - 47% by abundance, but that translated to only 0.01 – 1.4% of the total biovolume because of their small cell size. Large filamentous green algae are slower growing and occurred most often on the sides of stable cobbles where there is more protection from scour and shear. They colonized or drifted onto the artificial substrates during the 8 - 12 week fall and summer deployments, accounting for 0 – 44% of biovolume. Their prevalence was much lower in the winter at 2 – 7%. The nuisance diatom *Didymosphenia geminata* (Didymo) was detected at all LCR sample sites and was most prevalent in winter samples.

Despite the moderate and stable production in LCR in years with typical flows, there were substantial differences in the composition, abundance and biomass of periphyton observed between the three seasonal deployments in LCR. The periphyton community composition of winter showed differences from periphyton communities of fall and summer (Figure 3-8). Although these differences are small they are statistically significant (Appendix-A6). Other results suggest differences in the winter periphyton community composition. For example, winter has very little algal species delivered from the ALR. In contrast, fall and summer have a higher percent of their community from ALR algal species (Figure 3-9). Over the years of sampling, an average of 14±14% of the total periphyton in the summer and fall was attributable to reservoir phytoplankton, while only 2.1 ±1.7% was reservoir phytoplankton in the winter samples. Donations from the reservoir are therefore not responsible for the significant winter periphyton growth. Species richness and diversity (Simpson's index) did not have large variation between seasons. Average species richness ranged from 24 ± 4 to 46 ± 6 in LCR samples, with an overall average of 33 ± 4 taxa. Periphyton diversity in LCR was far higher than the diversity observed in MCR despite a similar range of substrates, indicating that LCR has better growing conditions, likely due to a combination of more stable flows, higher nutrients and higher water temperatures.

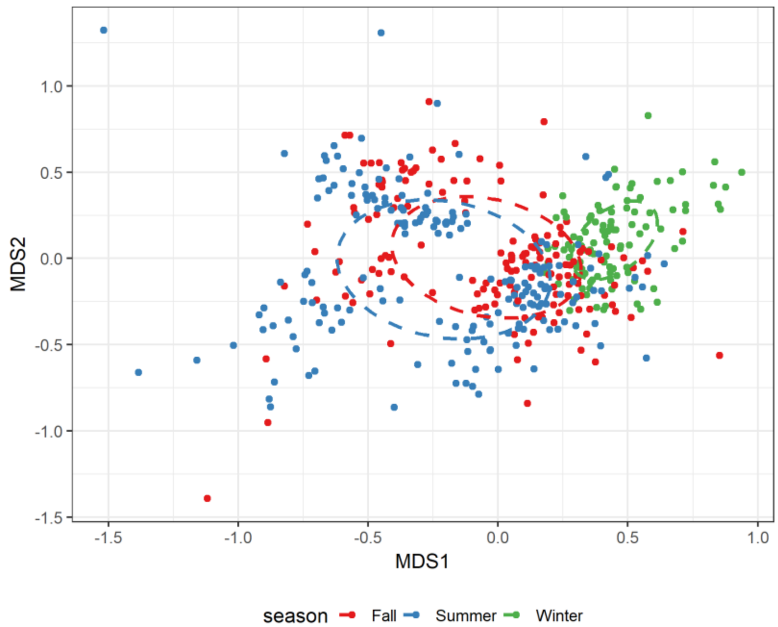

Figure 3-8: NMDS of periphyton genus level abundance grouped by season for all data from 2008 – 2016. The closer points are together the more similar the periphyton community composition is. The NMDS used a Bray-Curtis dissimilarity index and had a stress index of 0.20 (for more details see Appendix-A6). Ellipses are calculated based on 95% confidence interval of the NMDS scores for each group.

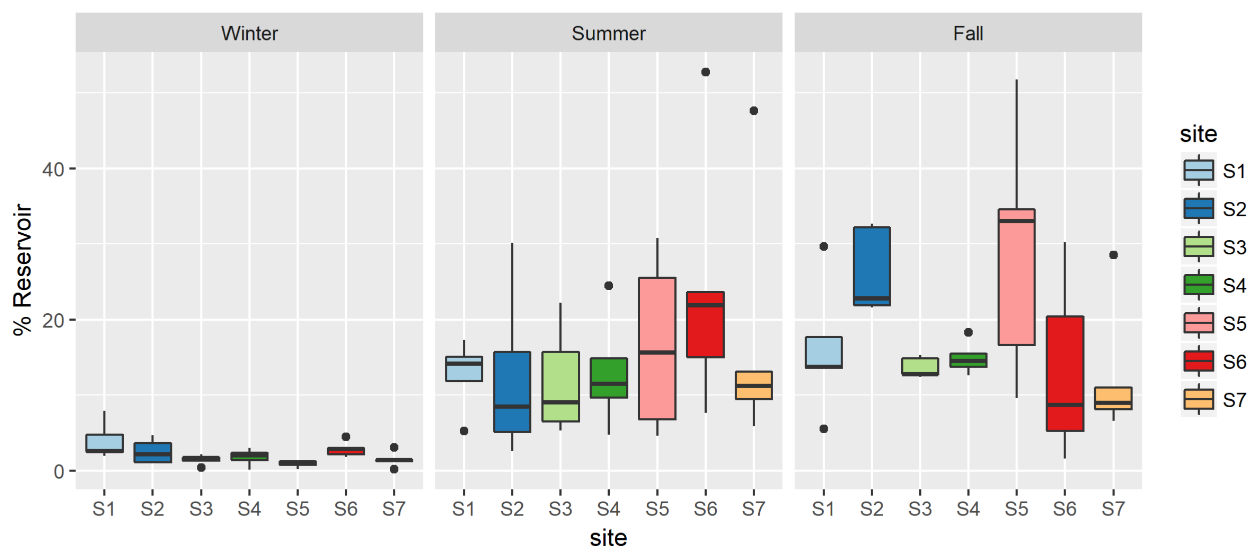

Figure 3-9: LCR periphyton community composition for 2016 showing the percent of species derived from ALR Reservoir. S=shallow, MS=mid-shallow. M=mid, MD=mid-deep, D=deep.

The periphyton production metrics of total biovolume, chl-a, and total abundance varied among seasons. The summer period which includes freshet flows had the lowest periphyton production across all years compared to other sample periods. The winter sampling period had higher periphyton biovolume compared to the summer and fall sampling periods (Figure 3-10). The biovolume results were affected by the higher occurrence of Didymo in winter. The winter and fall sampling periods had comparable chl-a and periphyton abundance. The highest values of chl-a in 2016 were seen in a few of the winter samples (chl-a>40 μg/cm^2). As expected, LCR production metrics for biovolume and chl-a were correlated ($r=0.59$, $p<0.001$).

The occurrence of the nuisance algae Didymo was highest in winter. To test if differences in winter flow conditions cause Didymo growth, a statistical CART model was run (Appendix-A6). The most important predictor of Didymo growth was low water temperature, followed by site. Generally, sites with back-watering from the Kootenay River (S4, S5, and S7) had higher relative abundance of Didymo than the sites S1, S2, S3, and S6Error! Reference source not found. These conditions provide favourable Didymo growth onditions - rocky substrates with cool, clear, moderate flows (Bergey et al. 2009; Bothwell et al., 2009).

Abundance, biovolume and chl-a consider live periphyton while ash-free dry weight (AFDW or volatile solids) includes all live and dead organic material. Like other metrics, AFDW analyses confirmed that winter is by far the most productive period in LCR for periphyton, and again, Didymo growth was a key driver (Figure 3-10 and Figure 3-11). Summer seasons with the freshet flow periods were consistently lowest for AFDW, likely due to substrate scouring during high flows. Fall results for AFDW can be inflated by caddisfly biomass that can exceed periphyton biomass, as in 2014.

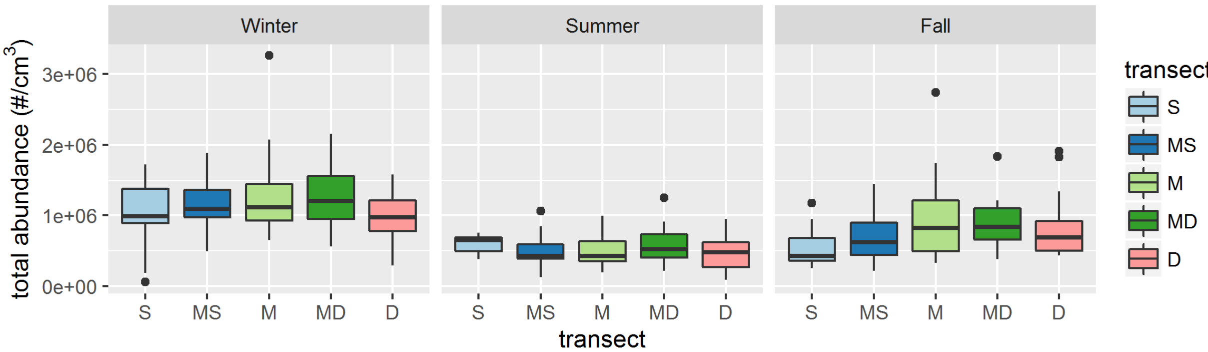

Figure 3-10: Periphyton abundance (cells/cm^2) biovolume (cm^3/m^2) and chl-a (µg/cm^2) in summer, fall and winter in 2014 and 2016, over the range of sampled depths. Depth labels are: S=shallow, MS=moderately shallow, M=mid, MD=moderately deep, D=deep.

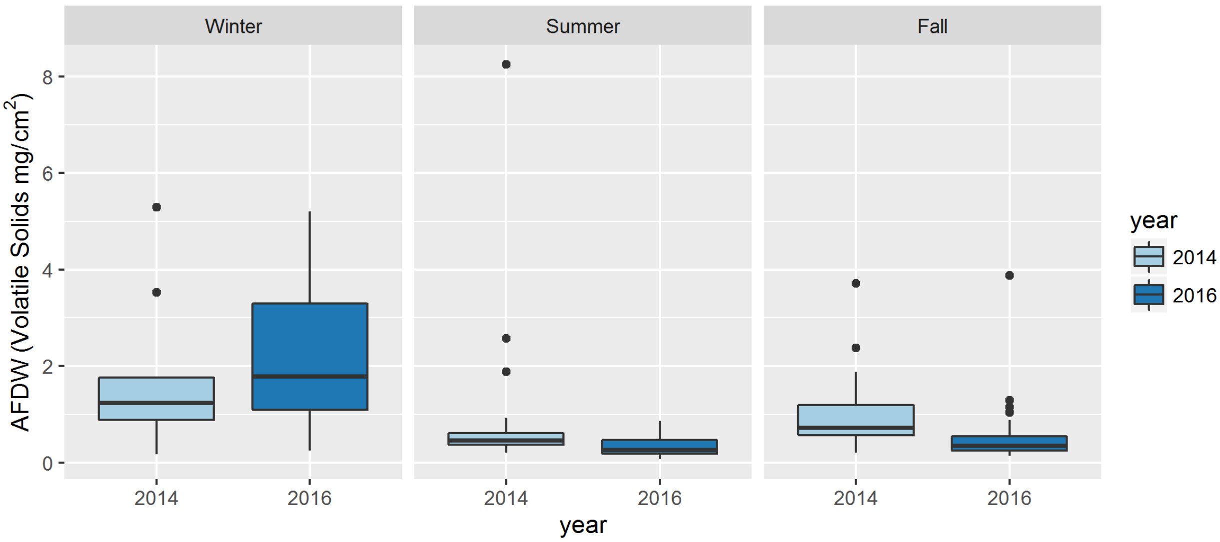

Figure 3-11: Ash-free dry weight (mg/cm^2) by season and year in 2014 and 2016

3.3.3 Evidence of Dewatering Substrates

There was evidence of periodic dewatering during the FFF period in 2016 on shallow and mid-shallow samplers. Sites with the most dewatering included S2, S3, S4 and S5. The number of hours that the samplers were dewatered ranged from 2 to 66 (Appendix-A7). Most shallow sites showed lower productivity and diversity during the 2016 fall sampling period. These shallow sites also showed the highest percent dead diatoms both in abundance and biovolume. For example, percent dead abundance decreased from 12% at shallow sites to 7.4% at deep sites (dead diatom biovolume decreased from 16.8% to 10.4%). Also, there was higher dead biovolume in 2016 with frequent substrate dewatering, compared to 2014 when there was less dewatering.

3.3.4 Periphyton Production Models

Periphyton mixed effects models were used to test if MWF, RBT, and FFF resulted in an increase of total biomass accrual. Velocity and substrate score were used in all models to account for physical differences between sites and transects. Elev Diff (MWF), Elev Diff (RBT), and Flow Daily SD were used as measures of flow variability for their respective flow periods. These measures of flow variability are highly correlated with other more general measures of flow. Therefore we could only include one flow variable predictor in each model, and thus we could not separate specific components of flow variability.

The seven sites sampled in LCR have unique habitat characteristics, these characteristics result in differences of periphyton production and community composition between sites. A summary of differences between sites is presented in Appendix-A8. For modelling purposes site is used as a random effect. Using site as a random effect controls for between site variations.

Some periphyton production models explained very little variation (R^2=0.12), whereas other models explained a moderate amount of variation (R^2=0.55) (Appendix-A8). Fall and summer models explained a greater variation of periphyton production metrics then winter models. This is largely due to their only being 3 years of winter data and 5 years of both summer and fall data. Due to the modest to moderate strength of the periphyton production models an important predictor does not mean a causal relationship with the predictor of interest and productivity.

Winter:

The periphyton production models for the MWF flow period suggest only chl-a is influenced by a measure of flow variability, the Elev Diff (MWF). The model for the periphyton production metric of chl-a was the only model that suggests an increase of flow variability also increases chl-a. The biovolume model was not interpreted because the model explained limited variation (R^2=0.12-0.17). The periphyton abundance model suggested flow variability had no association with abundance (Figure 3-12).

Summer:

During the summer (RBT flow period; includes freshet), the periphyton production models suggested Elev Diff (RBT) did not increase biovolume, abundance or chl-a. Velocity was the only predictor that was associated with biovolume and chl-a. The models suggest that sites with higher velocities have lower periphyton production. The periphyton abundance model did not have any important predictors (rvi<0.7).

Fall:

During the fall (FFF low period), Flow Daily SD was used as a measure of flow variability. The periphyton production models of biovolume and chl-a suggest Flow Daily SD has a negative association with periphyton production. Thus, an increase in flow variability results in a decrease of biovolume and chl-a. Similar to the summer abundance model, the fall abundance model did not have any important predictors (rvi<0.7).

Take Home:

Based on the data collected to date, the continued implementation of MWF flows is the only flow period that is potentially causing an increase of total biomass accrual of periphyton. Total biovolume and chl-a appear to be more sensitive to the MWF and FFF period then abundance. There is a known correlation between total biovolume and chl-a, so these results are not surprising. However, the models suggest that only MWF flows result in an increase of chl-a. During the RBT flow period the production models suggest that other factors are more important than flow variability. Fall periphyton production models of total biovolume and chl-a suggest flow variability decreases total biomass accrual.

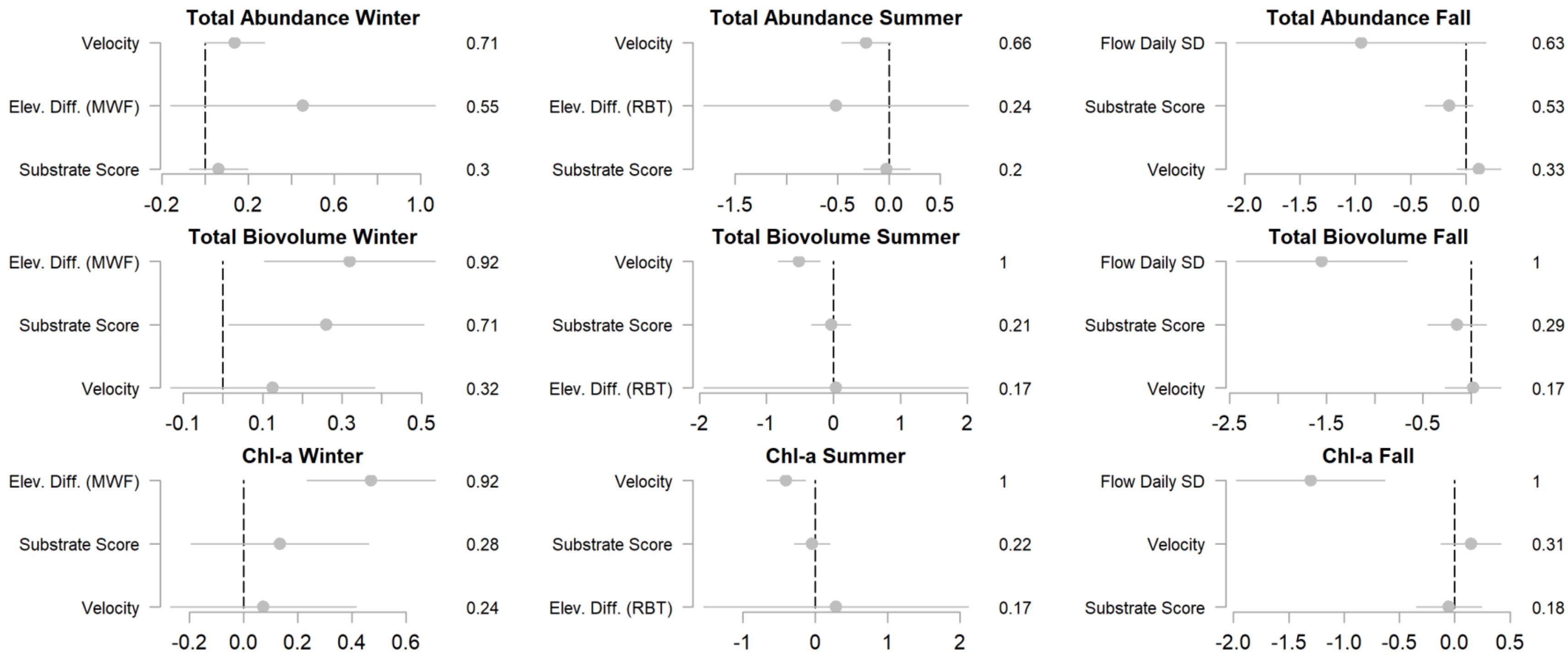

Figure 3-12: Mean coefficients and their 95% confidence limits of standardized explanatory variables of periphyton production in LCR (2009 – 2016). Coefficients are standardized to allow direct comparisons of the direction and size of effects, noting that variables with confidence limits that encompass zero can have either a positive or negative effect depending upon which model is considered. Key explanatory variables are sorted by their relative variable importance (RVI), values on the right-hand side y-axis of each panel.

3.4 Benthic Macroinvertebrates

Rock basket substrates for benthic invertebrates were used to address Ecological Productivity Monitoring Management Question 1, 2 and 3. During the three sampling sessions in 2016, 87% of rock baskets were recovered (Appendix-A9). Most of the loss occurred at shallow depths in the fall and was a result of the samplers being desiccated during substrate exposure.

3.4.1 Summary of Benthic Invertebrate Community Composition, Abundance, Biomass

As is usually the case, LCR had an abundant and diverse community of benthic macroinvertebrates in 2016. Rock basket sampling resulted in the collection of 40 different taxa in the fall, 43 in winter, and 48 different taxa in summer.

The 2016 benthic invertebrate data varied by season due to their life cycles. The highest mean abundance (#/basket) ± SD occurred in the summer with 12,568 ± 6,877 organisms per basket, followed by fall and winter with 7,599 ± 4,773 and 3,508 ± 3,765, respectively (Figure 3-13). Fall samples had the highest biomass, followed by summer, while winter was substantially lower (Figure 3-15). The winter 2016 biomass data had a similar range to the 2013 winter biomass data and both fell within the range of previous sampling periods. However, summer 2016 data had higher productivity compared to previous sampling periods (Figure 3-13 and Figure 3-15).

Mean species richness numbers were very similar across the three seasons, ranging from 20 ± 6 in the winter, to 24 ± 8 in the summer and 25 ± 6 in the fall. Dominant taxa in the summer and fall included Hydropsychidae (net-spinning caddisflies), and *Tvetenia (*non-biting midge; Chironomidae). In the winter, two taxa of Orthocladiinae (non-biting midge; Chironomidae) and Simuliidae (black fly) comprised approximately 83% of the samples. The dominant taxa sampled during each 2016 season were very similar to those documented in 2014. However, there was a higher relative abundance of Orthocladiinae in 2016 compared to 2013 and 2014. The shift in species abundance was also apparent in the relative biomass comparisons between seasons. Trichoptera was the dominant group in both the summer and fall, comprising 68.6 and 82.1 percent of the relative biomass, while they comprised only 14.8% in the winter. Gastropoda (57.9 %) and Diptera (14.3 %) maintained the greatest relative biomass in the winter. The decline in EPT and enhanced Chironomidae during the winter is likely a normal annual shift. A distinct winter benthic invertebrate community compared to summer and fall is further supported by the NMDS analysis (Appendix- A9 and Figure 3-14).

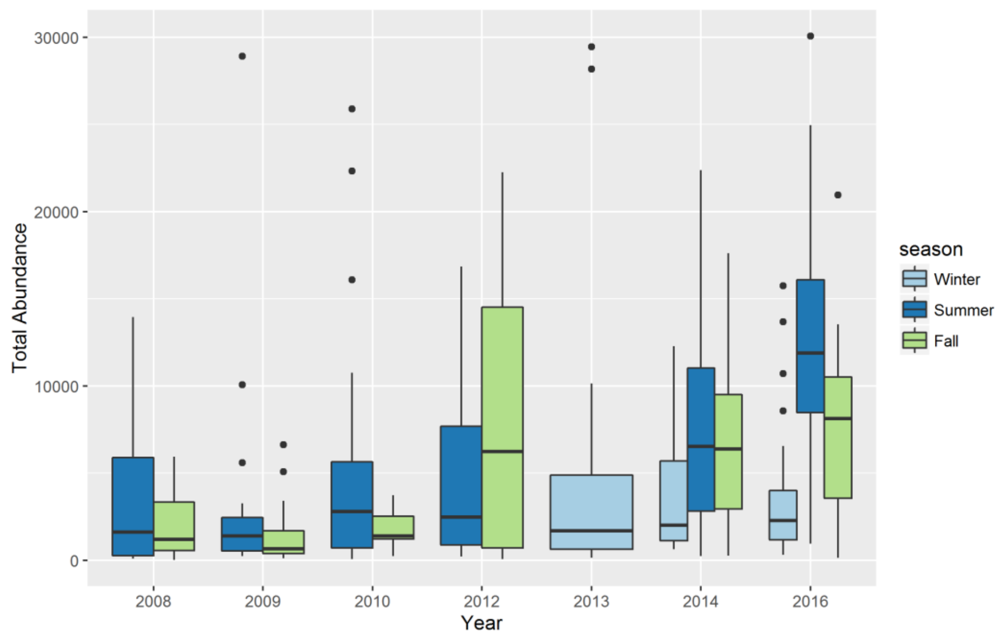

Figure 3-13: Total abundance of benthic invertebrates grouped by season and year.

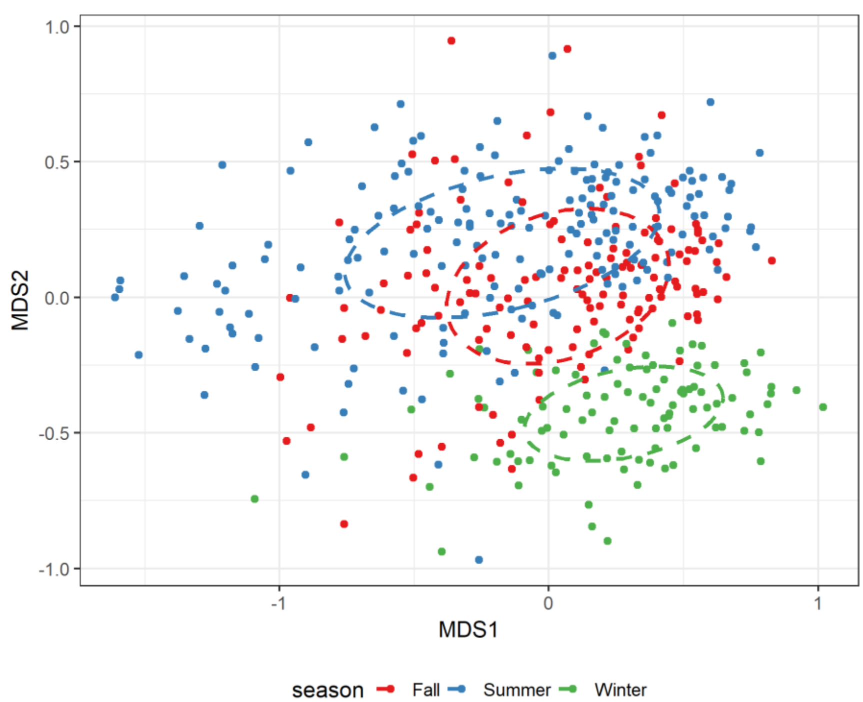

Figure 3-14: NMDS of invertebrate genus level abundance grouped by season for all data between 2008 – 2016. The NMDS used a Bray-Curtis dissimilarity index and had a stress index of 0.23. Ellipses are calculated based on 95% Confidence Interval of the NMDS scores for each group.

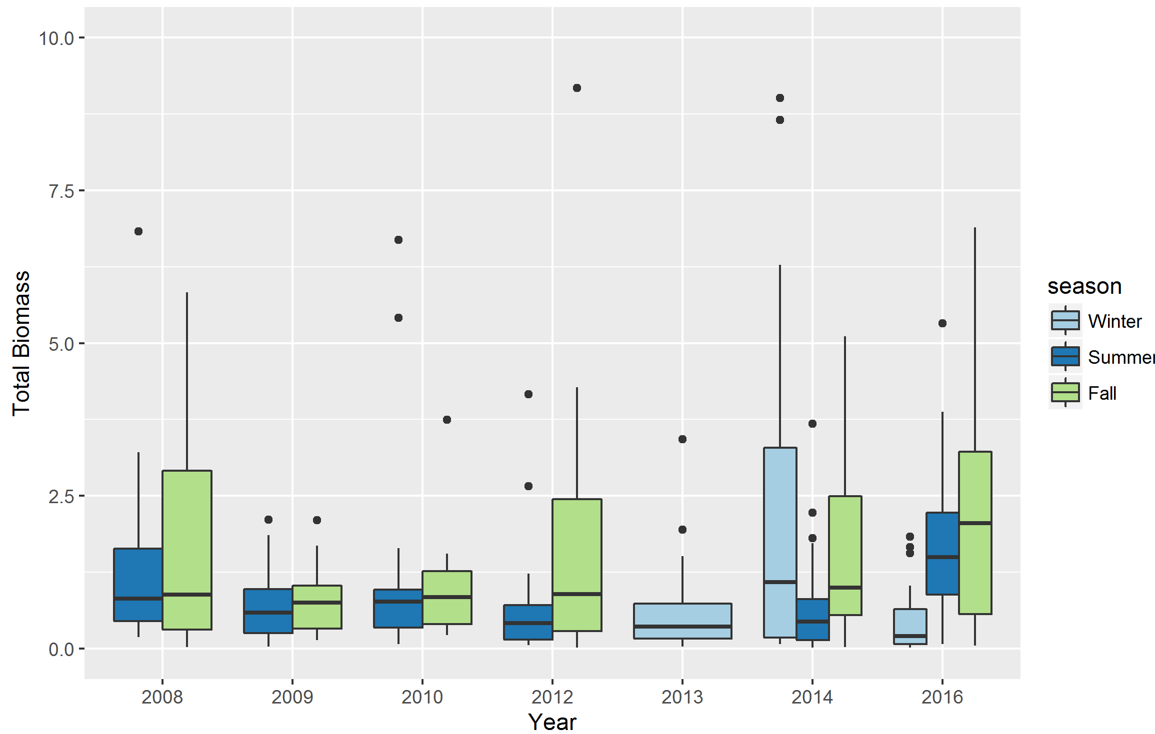

Figure 3-15: Total biomass (mg) of benthic invertebrates grouped by season and year.

Benthic Invertebrate communities can be assessed in terms of their value as food for fish, where Ephemeroptera, Plecoptera, Trichoptera (EPT) and Chironomids are good food. Percent EPT and Chironomidae populations have seasonal differences and patterns that are driven by their life histories. In the LCR, percent EPT is lowest in the winter and increases from summer to fall. In 2016, Percent EPT was 32±27% in winter and 61±28% and 75±30% in summer and fall, respectively. Percent Chironomidae had the opposite seasonal pattern, where percent Chironomidae was highest in winter, moderate in summer, and lowest in fall (Figure 3-16). Percent EPT was lower in winter 2014, compared to the winters of 2013 and 2016.

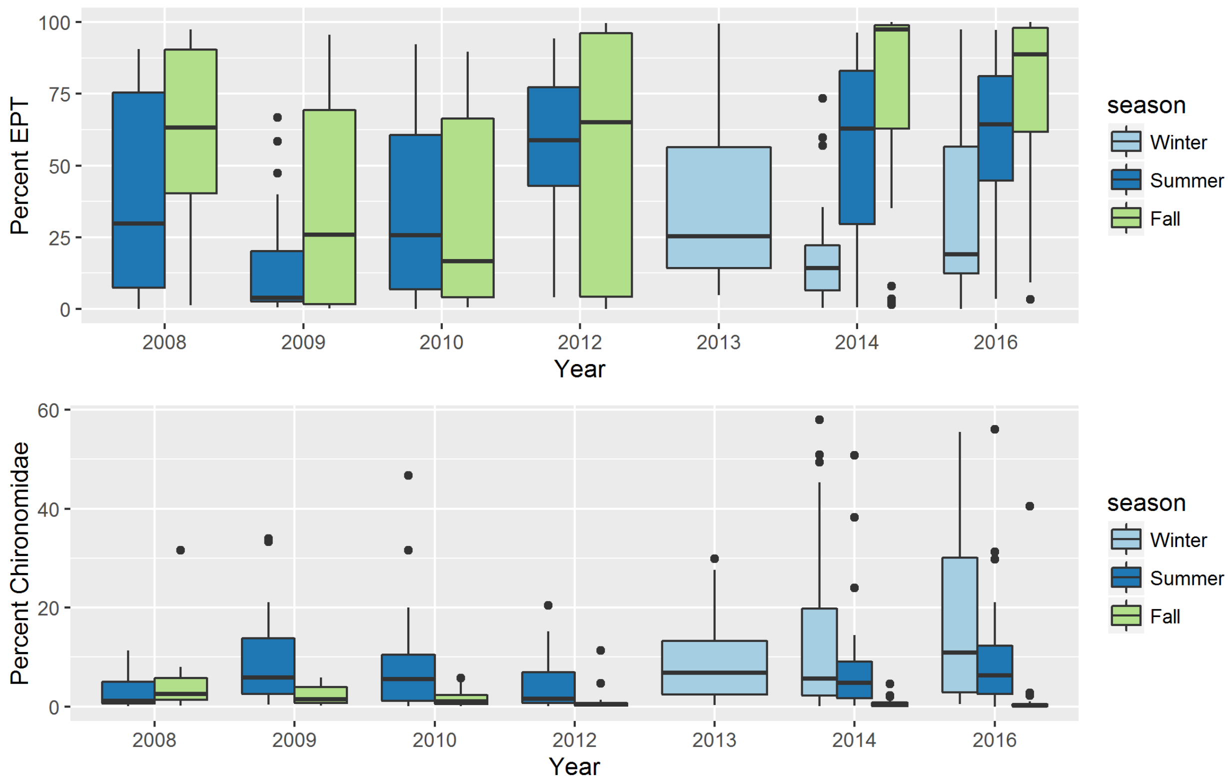

Figure 3-16: Percent EPT and Chironomidae of benthic invertebrates grouped by season and year.

3.5 Evidence of Dewatering Substrates

HLK flows were very low during the 2016 winter and fall deployment periods. During the fall retrieval, six of the fall samplers were completely dry. The temperature and light logger data confirmed that the shallow and mid-shallow samplers at S1, S2, S3, and S6 were dewatered for at least one day**.** These seven samples were not analyzed because they had minimal benthic invertebrate presence. Although some dewatering also occurred in winter, all winter samples were processed. S6-S had the lowest total abundance and highest species richness of all winter 2016 samples. This shallow sampler also had the highest estimated exposure time of 7 days.

3.5.1 Benthic Invertebrate Production Models

Some benthic invertebrate production models explained very little variation (R^2=0.08), whereas other models explained a moderate amount of variation (R^2=0.45), see Appendix-A10. Fall and Summer biomass and species richness models explained a limited amount of variation (R^2<0.20). Benthic invertebrate abundance models for all three seasons explained a moderate amount of variation (R^2=0.28-0.39).

Winter:

Elevation Diff (MWF), a measure of flow variability over the MWF flow period was used to test the effect of the implementation of MWF flows on biomass, abundance and composition of benthic invertebrates in the LCR. Winter benthic invertebrate production models included Elevation Diff (MWF), velocity, and substrate as predictors. The benthic invertebrate metrics of abundance and biomass had a positive association with velocity (Figure 3-17). However, the abundance and biomass models suggested flow variability had no association with benthic invertebrate production. Conversely, the models suggest flow variability had an effect on benthic invertebrate community composition, measured using the metrics species richness and Simpson's index. However, the benthic invertebrate community composition models explained limited variation (R^2=0.24-0.33). We therefore suspect that flow variability likely has a minor effect on benthic invertebrate community composition.

Summer:

Elevation Diff (RBT), a measure of flow variability over the RBT flow period was used to test the effect of the implementation of MWF flows on biomass, abundance and composition of benthic invertebrates in the LCR. Summer benthic invertebrate production models included Elevation Diff (RBT), velocity, and substrate as predictors. The benthic invertebrate models of community composition and production showed that velocity and to a lesser extent substrate score explained some variation in benthic invertebrate metrics (Figure 3-17). Flow variability was not an important predictor of benthic invertebrate community composition, biomass or abundance.

Fall:

Flow Daily SD, a measure of flow variability over the FFF flow period was used to test the continued fluctuations of flow on biomass, abundance and composition of benthic invertebrates in the LCR. Fall benthic invertebrate production models included Flow Daily SD, velocity, and substrate as predictors. Abundance was the only benthic invertebrate metric that showed an association with flow variability (Figure 3-17). The model suggests the greater the variation in flow the higher abundance of benthic invertebrates. The diversity of the benthic invertebrate community was only associated with velocity. The fall biomass and species richness models were not interpreted because these models explained a small amount of variation (R^2=0.08-0.17).

Take Home:

The continued implementation of MWF flows during the winter had an increased effect on benthic invertebrate community composition (e.g. species richness and Simpson's Index). Flow variability during the FFF period appeared to effect benthic invertebrate abundance, while managed flows during the RBT flow period were not an important predictor of the benthic community.

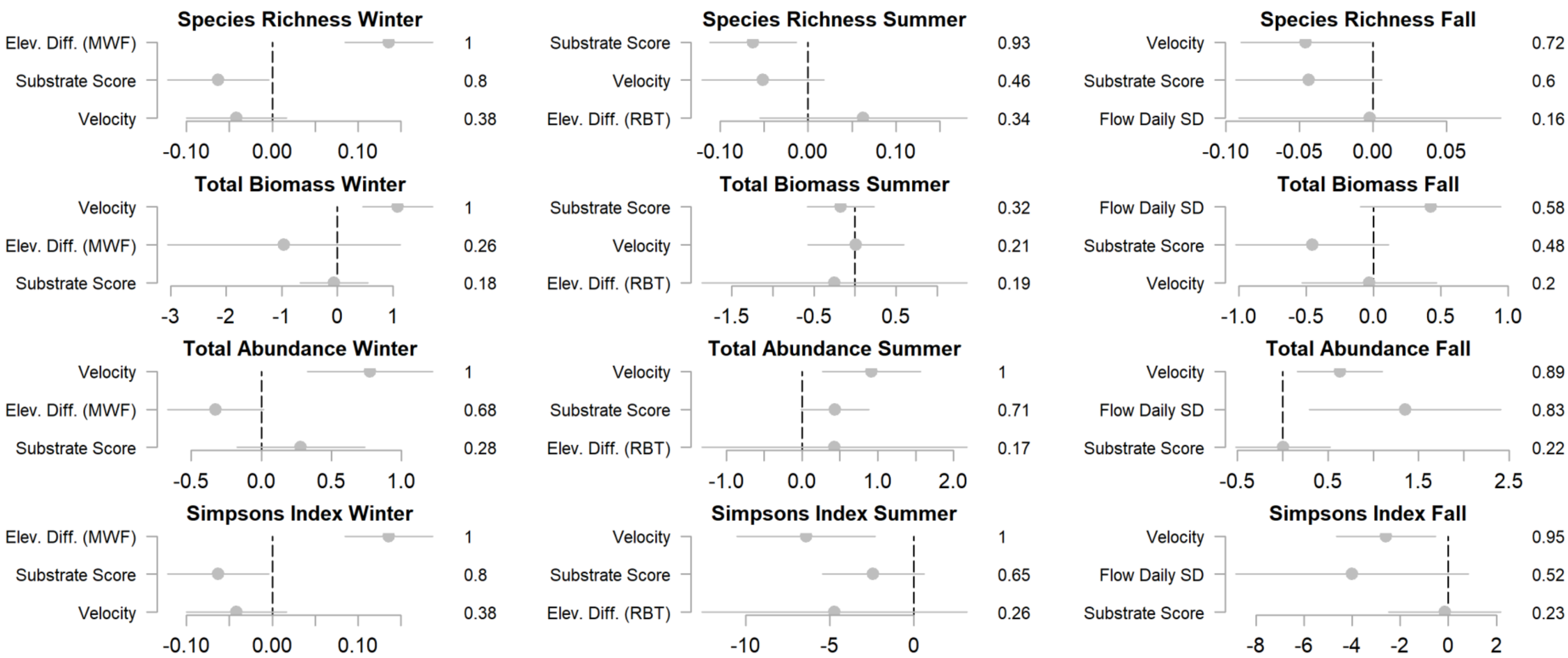

Figure 3-17: Mean coefficients and their 95% confidence limits of standardized explanatory variables of benthic invertebrate production in LCR (2009 – 2016). Coefficients are standardized to allow direct comparisons of the direction and size of effects, noting that variables with confidence limits that encompass zero can have either a positive or negative effect depending upon which model is considered. Key explanatory variables are sorted by their relative variable importance (RVI), values on the right-hand side y-axis of each panel.

3.5.2 Fish Food

The LCR rock basket samplers captured benthic invertebrates that are representative of diet data for both MWF and RBT (Golder 2009). Continued implementation of fish flows and their potential effects on the availability of fish food organisms was assessed to address $Ho3_{eco}$ (Figure 3-17). Percent quality forage and percent EPT are two indices used to measure the availability of fish food organisms. In previous years, benthic invertebrate families of Ephemeroptera (mayflies), Plecoptera (stoneflies), Trichoptera (caddisflies) and the order of Dipteran (true files) were assumed to be the benthic invertebrates MWF and RBT preferred to consume. To further explore diet preferences of MWF and RBT, the benthic invertebrate communities in fish stomachs were analyzed at the family level.

The stomach contents of 120 RBT and MWF caught in the fall of 2012 and 2014 were analyzed for benthic invertebrates. Seven of these fish had empty stomachs so were not included in the statistical analysis. The dominant taxa for most fish stomachs were Hydropsychidae (net-spinning caddisflies, Trichoptera). On average the percent relative abundance of Trichoptera in juvenile and adult MWF were 98±4.0% and 86±31%, whereas in juvenile and adult RBT the mean percent relative of abundance of Trichoptera were 64±37% and 64±35%. Although on average Trichoptera had the highest abundance in fish stomachs, there were some fish that had a higher abundance of Simuliidae (black fly; Dipteran). There were 2 adult RBT, 8 adult and 1 juvenile MWF that had greater than 50% relative of abundance of Simuliidae. Differences in fish stomach contents according to year, fish species and age were small and results are presented in Appendix-A11. There were a few fish caught in 2014 that had distinct community compositions. For example, the stomach contents of two adult RBT and one adult MWF caught in 2014 were dominated by Corixidae (water boatmen, Heteroptera). Two juvenile and one adult MWF, also caught in 2014, had stomach contents with the highest abundance of Glossosomatidae (little black caddisflies, Trichoptera).

The stomach content analysis confirm that fish consume Trichoptera and Dipteran which are included in the percent quality forage. In general, the dominant taxa in fish stomachs coincided with the most abundant benthic invertebrates during the fall sampling period. For example, in the LCR, Trichoptera made up 73% and 56% of the total biomass of benthic invertebrates in fall 2014 and fall 2012.

Percent EPT and percent forage quality models suggest that measures of flow variability are not the most important predictor of availability of fish food during all three flow periods. For the winter, summer and fall models velocity was the most important predictor of percent quality forage (Figure 3-18). The second most important predictor for the fall percent quality forage model was flow variability. The model suggested an increase in flow variability resulted in an increase of availability of fish food organisms. The percent EPT models for summer and fall had similar results to the percent quality forage models. In summer and fall, percent EPT and percent quality forage are highly correlated because the benthic invertebrate community is primarily composed of Trichoptera (caddisflies). In winter, percent EPT and percent quality forage are not correlated because the benthic community is primarily composed of Dipteran. The percent EPT model for winter explained limited variation and was not interpreted (R^2=0.14, Appendix-A10).

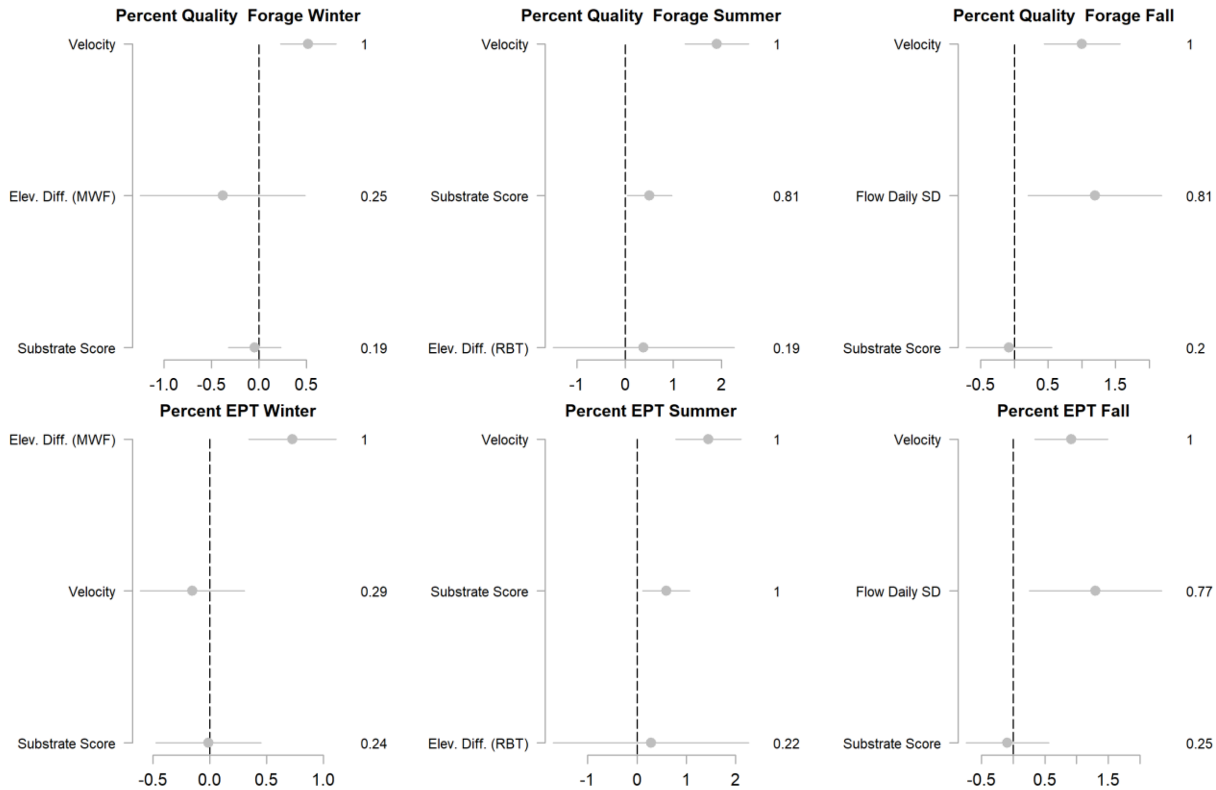

Figure 3-18: **Mean coefficients and their 95% confidence limits of standardized explanatory variables of benthic invertebrate fish food indices in LCR (2009 – 2016).** Coefficients are standardized to allow direct comparisons of the direction and size of effects, noting that variables with confidence limits that encompass zero can have either a positive or negative effect depending upon which model is considered. Key explanatory variables are sorted by their relative variable importance (RVI), values on the right-hand side y-axis of each panel.

4.0 DISCUSSION

4.1 Water Temperature

Water temperature varied seasonally, ranging from approximately 3.1 to 19.5°C and was generally consistent among years at the water quality index stations. The seasonal patterns observed were similar across all index stations, although the stations below the Kootenay River confluence were slightly warmer during the summer months. Given the baseline of released water temperature, LCR water temperatures were most influenced by air temperature, and upstream reservoir temperature. The data suggest that flow does influence water temperatures to some extent, but its effects are small compared to reservoir temperature and air temperature. The models determined that during the MWF and RBT flow periods, riverine temperature is more dependent on reservoir temperature than air temperature, whereas during the FFF period, air temperature is a more important factor.

We therefore continue to accept the null hypothesis Ho_{1phy} which states that continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not alter the seasonal water temperatures regime of LCR.

4.2 River Flows and Elevation

The 2016 freshet was lower and earlier than the previous five years. It peaked on June 14th, with approximately 3,677.9 m^3/s recorded at the Birchbank gauging station. Unlike previous years, the freshet peak occurred during the RBT protection flow period, which was designed to stabilize or increase flows from the beginning of April to the end of June to reduce redd dewatering and subsequent RBT egg losses (Baxter and Thorley 2010).

Historic water elevation data is not available, so a predicted data set was used to estimate water elevations prior to 2008. Since channel morphology has not significantly changed since 1984, a reasonably accurate prediction is possible because river elevation is a function of channel morphology. In wider channels, larger changes in flow are required to obtain the same changes in elevation when compared to narrow channels.

The modeling data indicate that both post-implementation (1995 – 2007) and continued (2008 – 2016) MWF flow periods resulted in smaller changes in water elevation between the spawning and incubation periods than pre-implementation of the flow regime (1984 – 1994). We expect reasonable strength in this relationship because predicted elevations were not different from those measured in the field for the period assessed. We therefore reject the management sub-hypothesis HO_{2Aphy}.

During the RBT flow period, the modeling data for WQIS2 and WQIS3 indicate that both the post- implementation and the continued RBT flow regimes caused a smaller cumulative decrease in river elevation than prior to the implementation of the flow regime. Like the MWF flow period analysis, modelled water elevations and those measured in the field were similar. We therefore reject management sub-hypothesis HO_{2Aphy}.

4.3 Water Quality

The hypotheses for water quality state that the continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall do not alter electrochemistry and biologically active nutrient concentrations in LCR. Testing of this hypothesis in 2014, concluded the tentative acceptance of the management hypotheses HO_{3phy}, HO_{3Aphy}, and HO_{3Bphy} (Olson-Russello et al. 2015). No further water quality sampling is planned for the remaining years of this contract.

The potential of other nutrients sources influencing LCR nutrients was investigated for this report. The correlation between DIN concentration at Arrow Lakes Station (ALR) 8 and DIN at WQIS1, indicate nutrient enhancement in ALR is likely influencing DIN in the LCR. However, the influence of nutrient enhancement on LCR was not detectable for T-P. Since both nitrate and phosphate are added to ALR, this result suggests that the phosphate was consumed by microflora within the reservoir. Although interesting, this analysis did not directly pertain to the influence of managed fish flows on water quality in LCR.

4.4 Periphyton Monitoring

The ecological monitoring management hypotheses HO_{2eco} states that the continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not increase total biomass accrual of periphyton in LCR. To address this management question, periphyton monitoring was undertaken during three separate sessions during the winter, summer and fall. Samplers were deployed in Reach 2 at seven sites and in the areas presumed to be most productive, ranging from the water's edge to 6 m deep.

4.4.1 Periphyton Production and Community Metrics

The LCR periphyton community is productive, diverse and variable. Most production metrics place LCR in the typical to productive range compared to other large rivers[3] (Table 4-1). Similarly, LCR periphyton diversity is stable and moderate relative to other large rivers. Despite high variability between seasons and years, the 2008 – 2016 accrual data indicate that peak biomass occurs in 6-7 weeks during summer, greater than 8 weeks in the fall and in greater than 10 weeks during winter.

Periphyton community compositions exhibited significant differences between years and seasons, and to a lesser extent between depth and site. Some of this variance may relate to flows and LCR operating regime, while some is likely attributable to variable nutrient and algal donations from the ALR, along with weather effects.

[3] Artificial substrates may inflate periphyton productivity metrics and we estimate the effect to be as large as 50% of the productivity on natural substrates (Larratt *et al.* 2013). However, our LCR data also suggest that periphyton metrics collected from closed cell Styrofoam substrate and data from stone tiles are similar, meaning it is reasonable to directly compare results from this study to the larger body of river periphyton research that utilized stone tiles.

Table 4-1: Summary of typical LCR periphyton metrics from 2008 to 2016, with comparisons to oligotrophic, typical, and productive large rivers and MCR

Metric	Oligo-trophic or stressed	Typical large rivers	Eutrophic or productive	MCR	LCR (median)
Number of taxa (live & dead)	<20 – 40	25 - 60	variable	5 - 52	8 – 60 (31)
Chlorophyll-a µg/cm^2	<2	2 - 5	>5 – 10 (30+)	0.04 – 4.1	0.01 – 55(3.6)
Algae density cells/cm^2	<0.2 x10^6	1 - 4 x10^6	>10 x10^6	<0.02 – 1.5 x10^6	0.03–3.9x10^6 (0.8x10^6)
Algae biovolume cm^3/m^2	<0.5	0.5 – 5	20 - 80	0.03 - 10	0.1 – 25 (3)
Diatom density frustules/cm^2	<0.15 x10^6	1 - 2 x10^6	>20 x10^6	<0.01 – 0.6 x10^6	0.06 – 6.94 x10^6 (0.81)
Biomass –AFDW mg/cm^2	<0.5	0.5 - 2	>3	0.12 – 4.8	0.04 – 8.3 (0.6)
Biomass –dry wt mg/cm^2	<1	1 – 5	>10	0.7 – 80	1.0 - 429
Organic matter (% of dry wt)		4 – 7%		1 – 10%	0.38 – 38.7 %
Bacteria sed. HTPC CFU/cm^2	<4 -10 x10^6	0.4 – 50 ×10^6	>50×10^6 – >10^{10}	0.2 – 5 x10^6	1.5 - >5 x 10^6
Fungal count CFU/cm^2	<50	50 – 200	>200	<25 – 600	8 - 1830
Accrual chl-a µg/cm^2/d	<0.1	0.1 – 0.6	>0.6	0.001 - 0.1 S 0.005 - 0.38 D	0.015 – 0.44 S 0.009 – 0.51 D

Comparison data obtained from Flinders and Hart 2009; Biggs1996; Peterson and Porter 2000; Freese *et al.* 2006; Durr and Thomason 2009; Romani 2009; Biggs and Close 2006.

Like all large rivers, diatoms dominated the LCR periphyton every year, along with variable contributions made by soft-bodied algae such as filamentous greens and cyanobacteria. The nuisance diatom Didymo was detected at all LCR sample sites and was most prevalent in winter. Species richness was lowest in the fall, particularly at the shallow sites that experienced dewatering and at deep sites where light penetration was low.

The periphyton community composition of winter was distinct from the periphyton community composition of summer and fall. The ALR donates very little phytoplankton in winter, compared to fall and summer. Reservoir periphyton contributions in summer and fall are significant and have been observed in other river systems immediately downstream of a reservoir (Truelson and Warrington, 1994; Bonnett *et al.* 2009). Differences in the periphyton community composition in winter, are likely a result of an increased presence of Didymo. Modelling confirmed that the relative abundance of Didymo was highest in winter. Winter is known as a stable flow period, a more stable pattern is thought to increase the success of the invasive Didymo (Bunn and Arthington 2002). Didymo prefers an oligo- to mesotrophic habitat with cool water, a stable flow regime with high exposure to UV-B radiation and cobble substrates. These ideal conditions are commonly located in lake-fed rivers, or in regulated rivers below reservoir impoundments such as LCR (Shelby 2006). Interestingly, modelling suggested site specific effects were more important than flows in predicting relative abundance of

Didymo. The sites with the highest amount of Didymo were furthest downstream from the HLK.

4.4.2 Influence of Managed Flows on LCR Periphyton Community

Periphyton in LCR showed significant variations in production and community structure between seasons and between years. Many factors that influenced periphyton production gradients are related to LCR flows contributed by reservoir releases. Our field observations agree well with the statistical models that measures of flow variability and velocity are the important factors influencing periphyton production in LCR. These results suggest a direct link between productivity and operations. Although discharge clearly plays a role on the periphyton community, the smaller nuances of managed fish flows (MWF, RBT and FFF) is more difficult to discern. Each managed flow period and relevant hypothesis is considered separately in the following sections.

4.4.2.1 Winter MWF Flow

Lower temperatures of 4 – 6^{o}C and reduced light intensity coupled with shorter day length apparently exerted less influence than the benefits of stable winter flows because winter samplers showed higher overall periphyton production than other flow periods, however, the time to achieve biomass was longer. Cool winter water temperatures will restrict growth of most green algae and some cyanobacteria, but not diatoms or most flagellates (Wetzel 2001), explaining the very low abundance of filamentous green algae in winter samples and the prevalence of low-light tolerant cyanobacteria, diatoms and Didymo.

The periphyton production models for the MWF flow period suggest chl-a is influenced by a measure of flow variability, the Elev Diff (MWF). The model for the periphyton production metric of chl-a suggested an increase of flow variability also increases chl-a. The biovolume model was not interpreted because the model explained limited variation (R^2=0.12-0.17), and the periphyton abundance model suggested flow variability had no association with abundance. Based on these findings, the continued implementation of MWF flows is the only flow period that is potentially causing **an increase of total biomass accrual of periphyton.** Based on these results, we tentatively reject hypothesis HO_{2Aeco} that MWF flows do not increase total accrual of periphyton or their biomass.

The model results for winter are still preliminary because there is only 3 years of data. In addition, there are extreme flow years within these year. For example, 2013 hard a large elevation difference between MWF spawning (Jan 1-21) and MWF incubation (Jan 21-Mar 31) and also had a high Flow Daily SD (Figure 4-1). It is suspected that 2013 data is amplifying the effects of Elev Diff (MWF) on periphyton production.

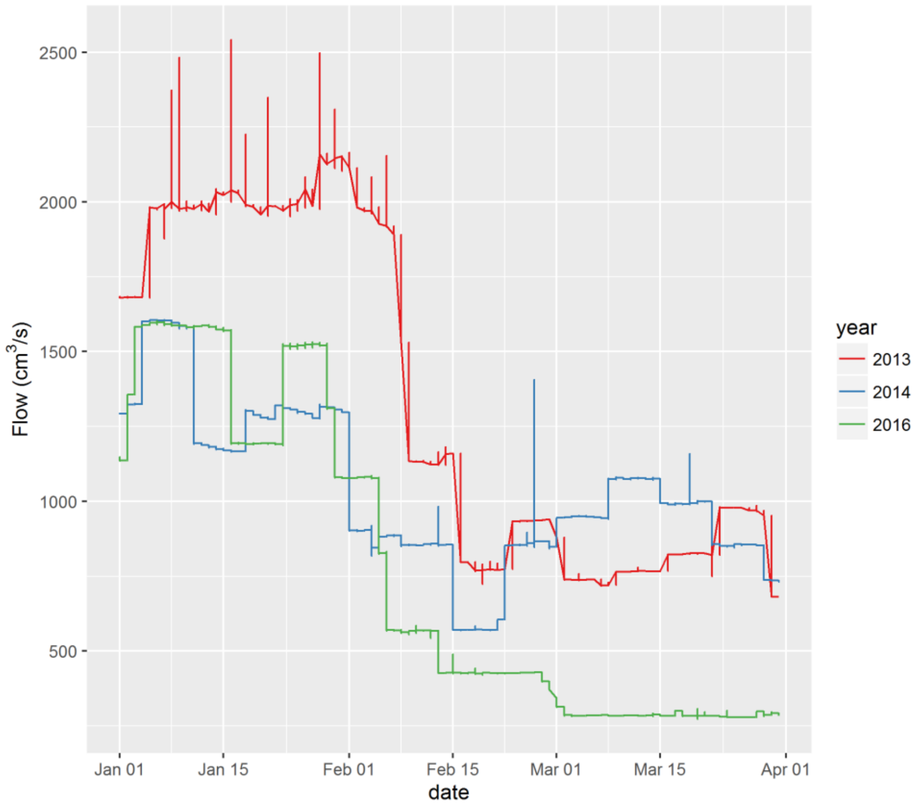

Figure 4-1: Hourly flows from HLK dam for the MWF flow period during winter sampling.

4.4.2.2 Summer RBT Flows

The lowest overall periphyton production and diversity were observed during the summer when freshet was occurring. Shear and scour of periphyton from higher velocities during high flow periods are likely the cause of this observation[4]. Reduced periphyton growth following high flow events is frequently observed in other river systems (Blinn *et al.* 1995, Biggs 1996, Bunn and Arthington 2002). Specifically, filamentous green taxa can be dislodge readily from the stream bed with small increases in velocity, while tightly attached diatoms require increased shear stresses to experience the same scour (Biggs 1996).

The high flows of freshet overshadow the managed RBT flows both in scale and apparently in effect on periphyton. The periphyton production models suggest that Elev Diff (RBT), which is the sum of the elevation drops during the deployment dates that coincide with the RBT flow period, did not increase biovolume, abundance or chl-a. Velocity was the only predictor that was associated with biovolume and chl-a. The models suggest that sites with higher velocities have lower periphyton production. Based on this, we tentatively accept the null hypothesis HO_{2Beco} that RBT flows do not increase total biomass accrual of periphyton in LCR.

[4] Increased flows do not always directly translate to increases in velocity, but generally, as flow increases, velocity also increases.

4.4.2.3 Fall Fluctuating Flows

The moderate flows during the FFF period allowed more periphyton growth compared to the summer, resulting in a relationship between season and production. Across all years, periphyton productivity increased during the fall at most sampled depths, except for several shallow sites. Periodic dewatering of shallow substrates along the water's edge reduced their fall periphyton production and increased mortality. In FFF periods with low flows, large areas of dewatered substrate occur and periphyton productivity shifts to deeper substrates, but may not fully compensate for the areal loss of shallow substrates. All wetted substrates also experienced variable scour as flows changed. A clear line of increased periphyton and filamentous green algae growth marked the position of the end of the varial zone and the beginning of the permanently wetted substrates developed each fall.

During the fall (FFF low period), Flow Daily SD was used as a measure of flow variability. The periphyton production models of biovolume and chl-a suggest Flow Daily SD has a negative association with periphyton production. Thus, an increase in flow variability results in a decrease of biovolume and chl-a. Based on this, we tentatively accept the null hypothesis HO_{2Ceco} that FFF do not increase total biomass accrual of periphyton in LCR.

4.5 Benthic Invertebrate Monitoring

The ecological monitoring management hypotheses HO_{1eco} states that the continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not affect the biomass, abundance and composition of benthic invertebrates in LCR. To address this management hypothesis, invertebrate monitoring was undertaken during three sessions in the winter, summer and fall. Samplers were deployed in Reach 2 at seven sites and in the areas presumed to be most productive, ranging from the water's edge to 6 m deep. The managed flow periods have been implemented in LCR long enough that resulting shifts in the benthic invertebrate community have likely stabilized (Poff and Zimmerman 2010). Six years of benthic invertebrate data was collected between 2008 and 2016, and no data was collected prior to the implementation of managed flows.

It is well known that flow is a major determinant of both physical habitat and biotic composition in rivers, and that aquatic species have evolved life history strategies that are in response to natural flow regimes (Bunn and Arthington 2002). Likewise, the effects of large impoundments on river ecology has also been well documented (Bunn and Arthington 2002; Konrad et al. 2011). The focus of this study is to understand if the managed fish flows have had an effect on the benthic invertebrate community, beyond what is typical given LCR's impoundment and deviation from a natural system. In both cases, the MWF and RBT flow regimes are geared towards reducing daily flow variability, which is likely more similar to a natural system.

4.5.1 Benthic Invertebrate Community Structure and Production

Table 4-2 provides a comparison of benthic invertebrates in other large river systems. The benthic community in LCR is remarkably more stable, diverse and productive than that of the Middle Columbia River. This is apparent when comparing the mean number of invertebrates per sample. The more consistent LCR flows appear to greatly benefit the benthic invertebrate community not only in abundance, but also in the prevalence of more sensitive, high quality fish food taxa such as EPT.

Despite the similarities of the annual LCR hydrograph to a natural system, hydrologic differences do exist. In other river systems, flow regulation has been shown to favour less sensitive invertebrate species such as orthoclad chironomids (Poff and Zimmerman 2010; Munn and Brusven 1991) and there is evidence of this in LCR. For example, chironomids are top contributors to relative abundance, especially in winter. An increased predominance of filter-feeding benthic invertebrates has also been documented in regulated river systems and LCR has high relative abundances of net-spinning caddisflies of the family Hydropsychidae during the fall and summer. These are good food for fish. Also, there is a lower abundance of Ephemeroptera (mayflies) which are sensitive to changes in flow (Szczerkowska-Majchrzak *et al.* 2014; Kennedy *et al.* 2016). Thus, in these aspects, the LCR benthic invertebrate community is typical of a regulated river system. However, given its ranking as a diverse and productive system, regulation has not resulted in obvious impairment of its benthic invertebrate productivity.

Coupled with the effects of regulation on the invertebrate community, other variables such as nutrient additions through the ALR fertilization program, industrial effluents (Celgar), municipal effluents, and invasive species (Didymo) all influence the overall distribution, abundance, and diversity of the LCR benthic community. This makes it difficult to separate the specific effects of a given flow regime from natural, annual and seasonal variation, and from variation originating from the influences of other ongoing factors (Bunn and Arthington 2002). Thus, specifically elucidating the effects of flow regulation, and more specifically the effects of managed fish flows from other stressors and inherent natural patterns on the benthic community has yet to be done with certainty.

Table 4-2: Comparison of benthic invertebrate communities in different river systems

River	Average Annual Discharge (m3/s)	Mean # of Invertebrates (±SE)	Total # of Taxa	Diversity (Simpson's Index)	Most Abundant Taxa (percent abundance)
MCR (Revelstoke)	955	278(±380)	27	0.48	Hydra sp. (43) Orthocladiinae (15) Orthocladius complex (9.4) Enchytraeidae (2)
LCR (Castlegar)-Winter	1,997	4541(±6379)	30*	0.7	Simulium spp. (29) Simuliidae (25) Orthocladius Complex (13) Orthocladiinae (9)
LCR (Castlegar)-Summer	1,997	6182(±6548)	51	0.78	Hydropsychidae (33) Hydropsyche (19) Tvetenia spp. (8) Simulium spp. (6)
LCR (Castlegar)-Fall	1,997	5278(±5391)	41	0.77	Hydropsyche (26) Tvetenia spp.(12) Tvetenia discoloripes group (9) Parachironomus (7)
Fraser River (Agassiz)	3,620	829 (±301)	55	0.84	Orthocladiinae (62.7) Baetis spp. (7.2) Ephemerella spp. (5.4)
Thompson River (Spence’s Bridge)	781	2108 (±1040.8)	48	0.44	Orthocladiinae (62.7) Baetis spp. (7.2) Ephemerella spp. (5.4)
Cheakamus River	–	1252 (±1149)	6	–	Ephemeroptera Plecoptera Diptera w/o chironomids

Data sources include Schleppe *et al.* 2013, Reece & Richardson 2000, Triton Environmental Consultants Ltd. 2008 and this report.
* 2016.R2-S6-S has been excluded from this calculation

Benthic invertebrate community structure in all rivers undergoes seasonal shifts in response to their life cycles (Giller and Twomey 1993). Chironomidae and EPT are indicator groups used to measure community balance. An even distribution of Chironomidae, Ephemeroptera, Plecoptera and Trichoptera usually indicates good biotic conditions. Populations with enhanced numbers of Chironomidae relative to EPT indicate environmental stress (Shelby 2006).

The LCR community composition of winter was distinct from the fall and summer sampling periods. Percent EPT was consistently lower in winter compared to other sampling seasons, while Dipteran Simuliidae tended to be higher during winter and similar seasonal shifts have been reported elsewhere (Giller and Twomey 1993). The decline in EPT and enhanced Chironomidae during the winter is likely a normal seasonal shift. The higher Gastropoda biomass seen in 2016 compared to the other winters may be a result of substrate dewatering that caused other, more mobile taxa to leave. In general, winter has lower species richness and diversity than summer or fall (Table 4-2).

Summer 2016 benthic abundance data were higher than previous summer sampling periods and may be a result of an earlier freshet. Fall 2016 data fell within the range of previous sampling results.

4.5.2 Winter MWF Flows

Elevation Diff (MWF), a measure of flow variability over the MWF flow period was used to test the effect of the implementation of MWF flows on biomass, abundance and composition of benthic invertebrates in LCR. Winter benthic invertebrate production models included Elevation Diff (MWF), velocity, and substrate as predictors. The benthic invertebrate metrics of abundance and biomass had a positive association with velocity. However, the abundance and biomass models suggested flow variability had no association with benthic invertebrate production. Conversely, the models suggest flow variability had an effect on benthic invertebrate community composition, measured using the metrics species richness and Simpson's index. Albeit, the benthic invertebrate community composition models explained limited variation (R^2=0.24-0.33). We therefore suspect that flow variability within the MWF flow period likely has a minor effect on benthic invertebrate community composition.

Based on this, we tentatively reject the hypothesis that the continued implementation of MWF does not affect the biomass, abundance and composition of benthic invertebrates in LCR.

4.5.3 Summer RBT Flows

Benthic invertebrate sampling did not completely overlap with the RBT flow period, but it did partially overlap during periods of increased flow associated with spring freshet. During this period, samplers were deployed and water levels subsequently increased, effectively altering "shallow" sites to more moderate depths over the duration of deployment. These once-shallow areas had increased biomass when compared to deeper areas in the river. From this, it appears that maintaining higher water levels during the RBT flow period, at minimum, stabilizes flows and limits the extent of desiccation events that negatively impact invertebrate and RBT redd survival. However, the larger effect of increasing freshet flows overshadows any possible benefit of the RBT flow operating regime. Statistical modeling data during the summer sampling period indicates that freshet is a predominant feature.

Elevation Diff (RBT), a measure of flow variability over the RBT flow period was used to test the effect of the implementation of MWF flows on biomass, abundance and composition of benthic invertebrates in the LCR. Summer benthic invertebrate production models included Elevation Diff (RBT), velocity, and substrate as predictors. The benthic invertebrate models of community composition and production showed that velocity and to a lesser extent substrate score explained some variation in benthic invertebrate metrics. Flow variability, associated with the RBT flow period, was not an important predictor of benthic invertebrate community composition, biomass or abundance.

Based on this, we tentatively accept the hypothesis that the continued implementation of RBT flows does not affect the biomass, abundance and composition of benthic invertebrates in LCR.

4.5.4 Fall Fluctuating Flows

Stable flows during the fall FFF period resulted in benthic community establishment that was similar to that of a natural system. Areas along the interface of the channel between the area of laminar flow and the channel edge were highly productive. Any effects of daily dewatering probably caused similar biomass loss to those documented in the Middle Columbia River (Schleppe *et al.* 2013), with the most significant influences occurring in areas that were frequently dewatered. Unlike previous years, shallow samplers were frequently dewatered in 2016 because flows steadily dropped throughout October.

Flow Daily SD, a measure of flow variability over the FFF flow period was used to test the continued fluctuations of flow on biomass, abundance and composition of benthic invertebrates in the LCR. Fall benthic invertebrate production models included Flow Daily SD, velocity, and substrate as predictors. Abundance was the only benthic invertebrate metric that showed an association with flow variability. The model suggests the greater the variation in flow the higher abundance of benthic invertebrates. The diversity of the benthic invertebrate community was only associated with velocity. The fall biomass and species richness models were not interpreted because these models explained a small amount of variation (R^2=0.08-0.17).

The effect of daily variability of flow on benthic invertebrate abundance provide evidence for the rejection of the hypothesis that the continued implementation of FFF does not affect the biomass, abundance and composition of benthic invertebrates in LCR.

4.6 Food for Fish

The fish stomach content analysis confirmed that RBT and MWF in LCR consumed primarily Trichoptera and Dipteran. Trichoptera and Dipteran are both included in the percent quality forage index. In general, the dominant taxa in fish stomachs coincided with the most abundant benthic invertebrates during the fall sampling period. For example, Trichoptera made up 73% and 56% of the total biomass of benthic invertebrates in fall 2014 and fall 2012, respectively.

Percent EPT and percent quality forage (EPT + Dipteran) models suggest that measures of flow variability are not the most important predictor of availability of fish food during all three managed flow periods. For the winter, summer and fall models velocity was the most important predictor of percent quality forage. The second most important predictor in the fall was flow variability. The model suggested an increase in flow variability resulted in an increase of availability of fish food organisms. Higher daily variability in flows may increase the shallow area that is available for EPT taxa to lay their eggs (Kennedy *et al.* 2016), and result in an increased abundance of EPT taxa. Based on this, we tentatively reject Ho3Ceco that continued fluctuations of flows during the fall do not increase availability of fish food organisms in LCR.

The percent EPT models for summer had similar results to the percent quality forage models. In summer, percent EPT and percent quality forage are highly correlated because the benthic invertebrate community is primarily composed of Trichoptera (caddisflies). We tentatively accept Ho3Beco because models suggest flow variability during RBT flow period did not increase the availability of fish food organisms (percent EPT and percent quality forage).

In winter, percent EPT and percent quality forage are not correlated because the benthic community is primarily composed of Dipterans. Winter models showed that percent quality forage is not affected by flow variability during the MWF flow period, while the EPT model could not be interpreted because it explained such limited variation. We therefore tentatively accept Ho3Aeco, because winter models suggest percent quality forage is not influenced by flow variability during the MWF flow period.

4.7 Summary and Next Steps

The literature clearly demonstrates that variables such as flow, velocity and substrates play a role in the overall characterization of the benthic community; this was further supported through our predictive modelling. However, the challenge occurs when trying to tease apart general flow variability from that of the MWF, RBT and FFF managed flows. To do this, we have attempted to develop predictive flow variables that are specific to each flow period. But the problem is that specific flow variables are highly correlated with overall flow and therefore it is impossible to determine if it is truly the managed flow that is having an effect and not just flow variability in general. To date, the data seems to indicate that when flows are high (e.g. during freshet), the effects of the managed flow period (RBT) is nominal, however in the fall and winter (FFF and MWF) when the flows are more stable, then the managed flow periods appear to play a larger role in shaping the overall benthic community. The goal of the remaining three years of this contract is to further explore better ways to isolate the true effects of each managed flow period from overall flow variability and to further improve our confidence in the roles of each managed flow period and how they may affect the individual metrics of the benthic community.

5.0 RECOMMENDATIONS

1. The final report of this study should focus on the detailed interactions between water quality, periphyton and benthic invertebrates using all previously collected data. Field sampling in 2018 will be consistent with that collected in 2016. Statistical analyses will be thoroughly explored to best understand the linkages between the various LCR components and the managed flow periods (MWF, RBT and FFF).

2. A data sharing agreement with Columbia River Integrated Environmental Monitoring Program (CRIEMP) should be established. Data from other LCR projects such as Celgar's EEM will help to investigate how this project's LCR sites differ from nearby sites. This investigation will be particularly useful to compare the S6 site to other depositional sites.

3. A coordinated effort between projects could help to develop more explanatory variables that could be used in future modelling. Ecoscape has had initial conversations with Poisson Consulting to investigate methods used to directly test management questions using various explanatory variables such as those that we have created (e.g., mean daily standard deviation in flow). Utilizing a consistent set of explanatory variables may help to better link productivity related effects to other ongoing studies such as fish indexing.

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Appendix A – Supplemental Results

Appendix-A1: River Flows Supplemental Results

Table A1: Mean Daily Flows in 2016 by Designated Flow Period (m3/s)

Year	Statistic	HLK/ALGS	Brilliant	Birchbank
Mountain Whitefish Flows (Jan 1 - Mar 31)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	90	90	90
	Minimum	279.0	371.8	923.2
	Maximum	1596.6	845.8	2037.5
2016	Median	428.4	517.0	1150.6
	Arithmetic Mean	763.7	580.3	1386.7
	Standard Deviation	518.1	158.2	393.4
	Coefficient of Variation	0.68	0.27	0.28
Rainbow Trout Flows (Apr 1 to Jun 30)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	90	90	90
	Minimum	284.4	789.6	1147.9
	Maximum	1836.5	2018.0	3142.7
2016	Median	427.1	1674.9	2480.7
	Arithmetic Mean	828.1	1493.6	2414.5
	Standard Deviation	530.9	407.6	580.3
	Coefficient of Variation	0.64	0.27	0.24
Fall Fluctuating Flows (Sep 1 to Oct 31)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	60	60	60
	Minimum	282.3	315.8	956.2
	Maximum	1565.3	937.2	2151.5
2016	Median	964.9	506.1	1411.3
	Arithmetic Mean	1010.9	506.9	1577.5
	Standard Deviation	471.2	164.6	403.1
	Coefficient of Variation	0.47	0.32	0.26

Table A2: Mean daily river flows (m^3/s) at HLK Dam, Brilliant Dam and the Birchbank gauging station in 2016

Location	N (days)	Statistic	2016
HLK	365	Mean	989.6
		Min	144.8
		Max	2163.0
		SD	627.5
Brilliant	365	Mean	884.3
		Min	315.8
		Max	2018.0
		SD	469.1
Birchbank	365	Mean	1943.0
		Min	923.2
		Max	3142.7
		SD	656.2

Appendix-A2: Water Levels Supplemental Results

The best models varied among the five WQIS sites and contained different sets of explanatory flow variables. The variance in elevation described by top models was typically very high (R^2 range: 0.89-0.98), suggesting that the use of these models for predictive purposes is plausible (Table A-3). The accuracy of the predictive elevations is further supported when the actual elevation differences during the post implementation period are compared to the observed elevations.

Table A3: The best fit models for each water quality index station that were used to predict historic water levels for the MWF and RBT flow periods

Site	Best Fit model (Intercept + Coefficient(±SE))	Adjusted R^2	p-Value
WQIS1	417.7 + BRD(-0.000496 ± 4.89e-05) + BRD^2 (3.69e-07 ± 2.24e-08) + HLK(0.00261± 5.17e-05) + HLK^2(-3.28e-07 ± 1.97e-08)	0.932	< 0.001
WQIS2	417.2 + BIR(2.96e-04 ± 3.35e-05) + BIR^2(-4.40e-08 ± 7.55e-9) + BRD(-4.00e-04 ± 3.22e-05) + BRD^2(3.27e-07 ± 1.33e-08) + HLK(2.84e-03 ± 4.06e-05) + HLK^2(-4.23e-07 ± 1.61e-08)	0.969	< 2.2e-16
WQIS3	416.6 + BIR(2.76e-04 ± 3.00e-05) + BIR^2(-2.35e-08 ± 6.86e-09) + BRD(-2.93e-04 ± 2.84e-05) + BRD^2(3.65e-07 ± 1.19e-08) + HLK(2.01e-03 ± 3.73e-05) + HLK^2(-1.98e-07 ± 1.48e-08)	0.975	< 2.2e-16
WQIS4	409.8+ BIR(1.78e-03 ± 9.36e-05) + BIR^2(-2.09e-07 ± 2.09e-08) + BRD(8.40e-04 ± 8.90e-05) + HLK(7.61e-04 ± 1.11e-04)+ HLK^2(-1.26e-07 ± 4.36 e-08)	0.889	< 2.2e-16
WQIS5	409.2+ BIR(4.09e-04 ± 3.33e-05) + BIR^2(-6.93e-08 ± 6.35e-09) + BRD(1.11e-03 ± 3.66e-05) + BRD^2(1.05e-07 ± 1.29e-08) + HLK(9.36e-04 ± 4.57e-05) + HLK^2(1.63e-07 ± 1.75e-08)	0.976	< 2.2e-16

NOTE: BIR = Birchbank BRD = Brilliant Dam HLK = Hugh L. Keenleyside Dam

MWF = Mountain Whitefish flows RBT = Rainbow Trout flows

Appendix-A3: Water Temperature Supplemental Results

The water temperature models contained all combinations of explanatory variables. There was only one plausible model for the MWF and RBT flow periods and eight plausible models (Δ AICc < 3) for the FFF period. The models explained a high proportion of the variance in LCR water temperature (R^2 = 0.57 – 0.62). Not surprisingly, LCR water temperatures were most strongly correlated with Castlegar air temperature and with ALR reservoir water temperatures when all flow periods were considered (Figure A1). Since the models described a high proportion of the variance, and model selection resulted in only one model being selected for two of the three flow periods, we conclude that the identified factors are key parameters affecting river temperature.

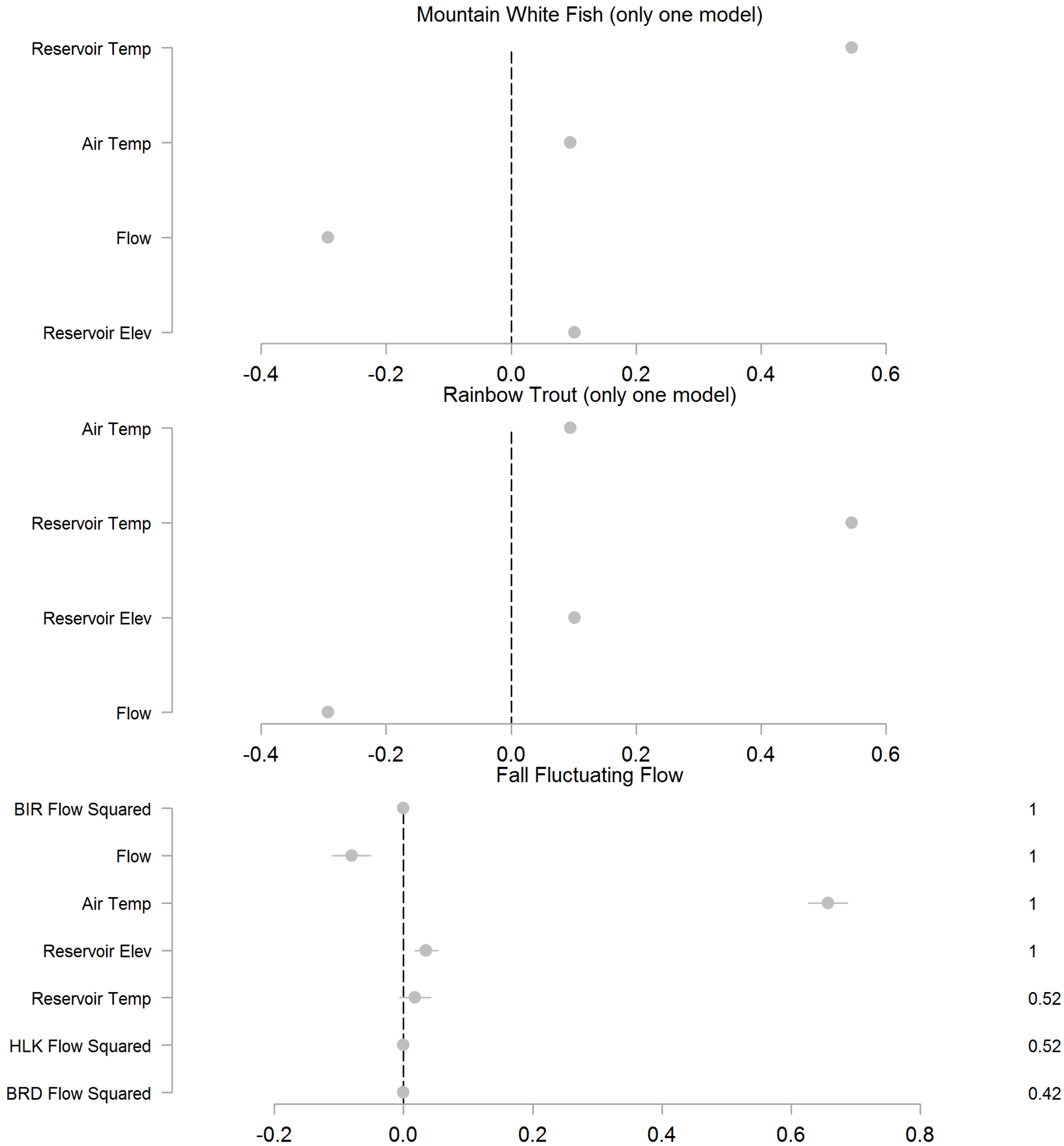

Figure A1: Scaled and Centered Parameter Estimates (circles) with 95% Unconditional Confidence Intervals (lines) from Averaged Predictive Linear Mixed-effects Models of LCR Water Temperature. Coefficients are standardized to allow direct comparisons of the direction and size of effects, noting that variables with confidence limits that encompass zero can have either positive or negative effects depending on which model is considered. Key explanatory variables are sorted by their relative variable importance (values on the right hand side of each figure).

Appendix-A4: Water Quality Supplemental Results

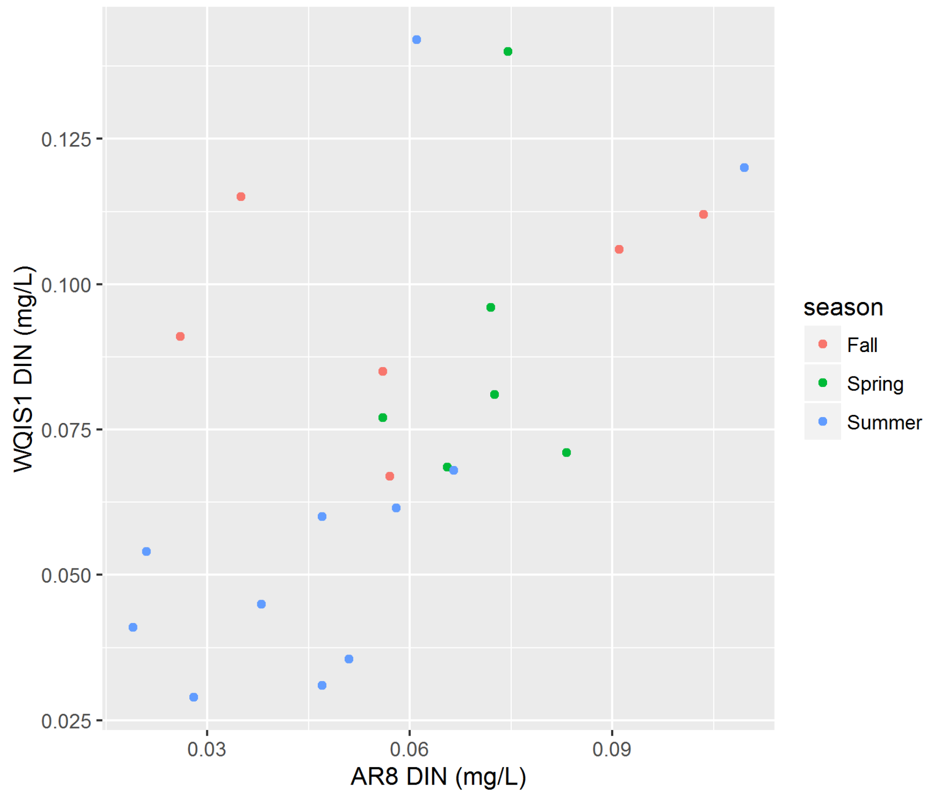

Figure A2: Average monthly DIN for 2008-2014 at WQIS1 and station AR-8.

Appendix-A5: Periphyton Accrual Supplemental Results

Table A4: LCR periphyton metrics for mid-depth samplers deployed for 10, 12, 20 and 26 Weeks in Winter 2013, 2014 and for 10 Weeks in Winter 2016

Winter Deployment Duration/Year	Chlorophyll-a $\mu g/cm^2$			Biovolume cm^3/m^2		
Sampler Depth	MS	M	MD	MS	M	MD
12 weeks in 2013	10.8	10.9	n/s	14.7	23.6	n/s
26 weeks in 2013	8.54	5.88	n/s	38.5	43.1	n/s
10 weeks in 2014	7.6	7.36	n/s	9.48	7.9	n/s
20 weeks in 2014	2.83	4.79	n/s	3.46	3.25	n/s
10 weeks in 2016	6.66	7.10	7.35	13.0	15.3	13.0

Time series accrual data for chl-a was collected in Winter 2014 and 2016. Both winters gave very similar results, with slow, steadily increasing chl-a through all ten weeks to high overall production. This chl-a data indicates that the accrual time required for LCR periphyton to reach peak biomass in the winter exceeds 10 weeks during stable flows. Both winter 2014 and 2016 could not be fit to a sigmoidal curve, further suggesting that LCR periphyton biomass is still increasing at 10 weeks.

Appendix-A6: Summary of Periphyton Community Composition and Productivity in the LCR Supplemental Results

Table A5: Range of Periphyton Relative Abundance and Biovolume Obtained from Artificial Substrates by Season and Year (averaged over sampler depth)

LCR	Summer 2008 – 2010, 2012, 2014, 2016		Fall 2008 – 2010, 2012, 2014, 2016		Winter 2013, 2014, 2016	
Algae Type	**Abundance (cells/cm^2)%**	**Biovolume (cm^3/m^2)%**	**Abundance (cells/cm^2)%**	**Biovolume (cm^3/m^2)%**	**Abundance (cells/cm^2)%**	**Biovolume (cm^3/m^2)%**
Diatoms	43 - 99	56 - 95	46 - 98	63 – 99	68 - 95	91 - 98
Flagellates	0.06 - 43	0.1 – 1.8	0.13 – 2.7	0 - 67	1.2 – 4.6	0.14 – 0.62
Cyanobacteria	0.6 – 42.5	0.1 – 0.59	1.3 – 46.5	0.02 -0.32	0.03 – 25.8	0.02 – 1.4
Green	0.44 – 9.8	4.8 - 43	0.7 – 5.5	1.1 - 34	0.26 – 2.0	1.9 – 6.8

Community analyses of the 2008 – 2016 periphyton data were completed at the genus level to allow focus on large-scale trends. The stress index was 0.20, which indicates the two NMDS axes are an adequate representation of the periphyton community composition. A permutational MANOVA indicated that periphyton community compositions exhibited significant differences when grouped season ($F=34.7$, $p<0.001$), season explained 14% of the variation in the periphyton community.

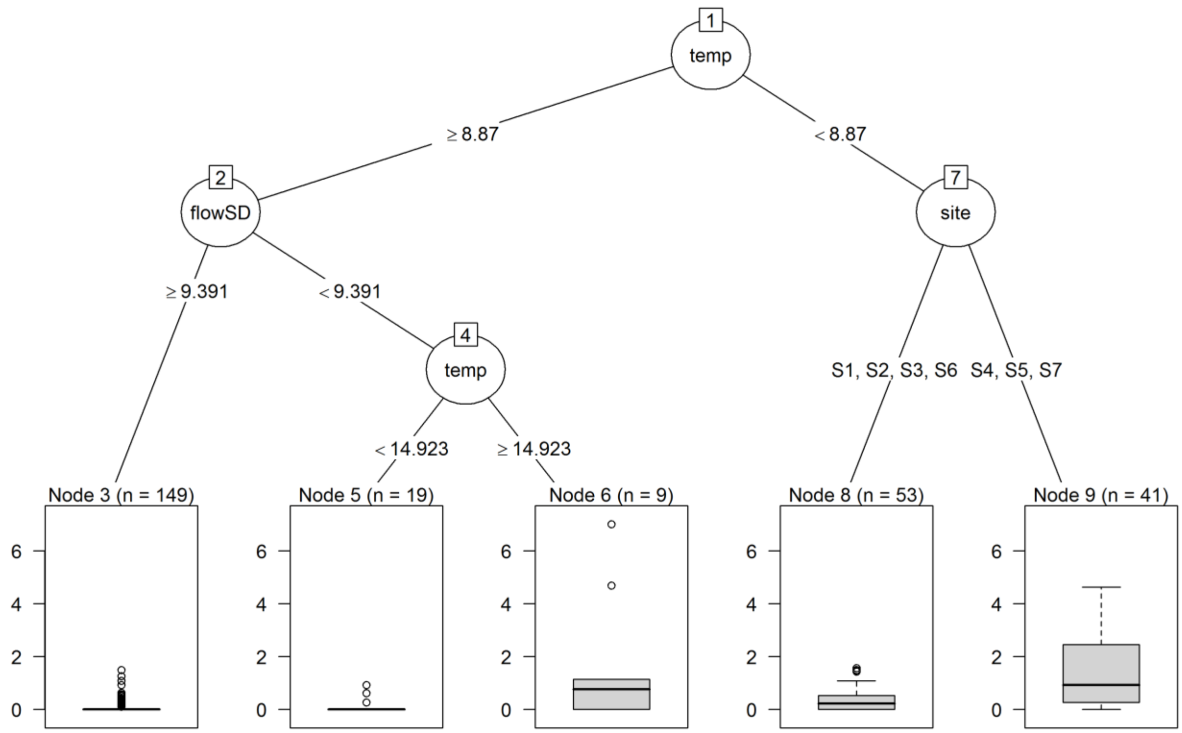

Figure A3: CART model for relative abundance of Didymo for fall, summer, and winter 2008-2016. The splitting rules indicate how the data is being grouped. The final groups (terminal nodes of the tree) show boxplots of percent relative abundance of Didymo. Temp is mean daily water temperature of the sampling period.

Appendix-A7: Evidence of Dewatering Substrates Supplemental Results

Table A6: Summary of artificial substrates estimated exposure times and productivity metrics in Fall 2016.

Fall 2016 data		production metric		
LCR Sampler	time exposed (hours)	abundance (cell/cm2)	biovolume (cm3/m2)	chl-a (ug/cm2)
R2 S1 MS	25	1.24	435596	1.15
R2 S2 M	31	1.39	332948	NA
R2 S2 MS	49	0.90	217140	NA
R2 S2 S	54	2.37	689584	NA
R2 S3 MS	52	1.46	276360	0.68
R2 S3 S	56	0.50	252672	0.36
R2 S4 MD	2	4.29	1055432	2.50
R2 S4 S	43	2.43	568512	0.39
R2 S5 S	15	1.37	671160	1.58
R2 S6 MS	51	1.40	280308	NA
R2 S6 S	66	2.16	422436	NA
R2 S7 S	6	0.86	397432	1.50

Appendix-A8: Periphyton Production Models Supplemental Results

When the 2016 data for the three LCR flow periods are considered, it is clear from the strong correlation between biovolume and chl-a that winter is a very productive season (Figure A4). 2016 periphyton data indicated that S7 had the highest productivity of the sample sites, likely due to moderate flows over cobble substrates. The only true depositional site S6 has high productivity, particularly in the fall. Of the erosional or mixed-character sites, S3 and S4 show the most consistent productivity, while S2 has the lowest overall productivity, likely due to substrate exposure during low river water levels.

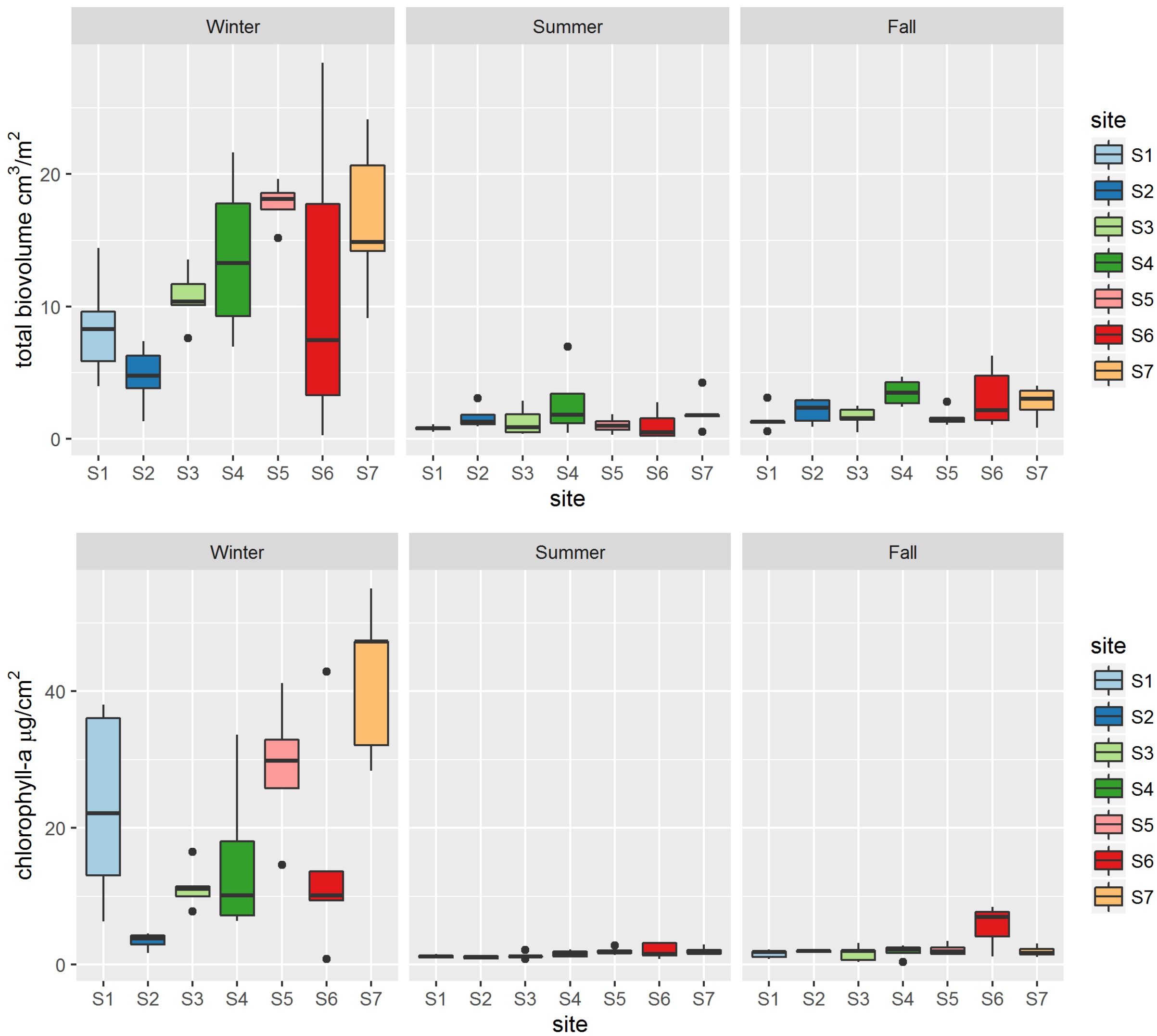

Figure A4: Periphyton biovolume and chlorophyll-a by site in the winter, summer and fall of 2016. Sites were classified by substrate type: Erosional S1, S2, S7; Depositional S6; Depositional during low flows (Sites 3, 4, 5).

Erosional fast-flowing sites with cobble substrates (S1 S2 S7) grew rapid colonizing diatoms with firm attachment strategies, while the lower velocity depositional and mixed sites (S6; S3 S4 S5) included more decomposer bacteria, detritus and motile species that can re-position their cells as sediments deposits. On gradually sloped cobble/gravel bars, a clear line of increased periphyton growth marked the end of the varial zone with periodic exposure, and the beginning of the permanently wetted substrates. This was similar to the banding patterns of biofilm observed on the MCR. Filamentous green algae never occurred on substrates that were periodically exposed. Their growth was greatest in moderate-low flow periods and at 1-2.5 m depths (MS samplers). The position of the interface between the main cool river flow and the shallow, slightly warmer back-eddy zones is directly related to flow and it appears that filamentous green algae distribution was influenced by this interface in the river.

Erosional habitats dominate in LCR and sample sites located in them (S1, S2, S7) had very high winter production, high fall productivity and moderate summer productivity (Figure A4). Depositional sites are less common in LCR; Site 6 was productive in the winter and fall seasons but was less so in the summer. During the winter deployment, shallow samplers in depositional areas were often partially buried in deposited sediments or decaying Didymo masses, both of which reduced periphyton growth. Viable Didymo mats were rarely encountered on the depositional substrates in any season, rather, they occurred on cobble substrates that experienced low velocity flows.

Periphyton production in erosional and depositional sites was similar in the summer but depositional sites had more chl-a in the fall, while most erosional sites had more productivity in the winter. These results agree well with the statistical models that consistently identified velocity, flows and flow variability as the most important factors influencing periphyton production in LCR. Results, specifically the autotrophic index, suggest that more photosynthetic production occurred at erosional sites, while more heterotrophic decomposition occurred at depositional sites. Chlorophyll-a and AFDW can be combined as a ratio into the autotrophic index (AI) = AFDM (in mg/m^2) / chlorophyll a (in mg/m^2) (Weber 1973). The autotrophic index is indicative of the proportions of the periphyton community composed of heterotrophic (fungi, yeasts, bacteria, protozoa) and autotrophic (photosynthetic bacteria and algae) organisms (Biggs and Close 1989; APHA 1995; Biggs and Kilroy 2000; Yamada and Nakamura 2002; Runion 2011).

Table A7: Summary of the Number of Plausible Models Identified using Model Averaging (those with a AIC <3) and the Range of Pseudo R^2 Values for Selected Periphyton Models

Periphyton Response	Winter		Summer		Fall	
	# of plausible models	range of pseudo R^2	# of plausible models	range of pseudo R^2	# of plausible models	range of pseudo R^2
Abundance	8	0.38-0.42	4	0.31-0.33	8	0.52-0.55
Biovolume	4	0.12-0.17	3	0.39-0.40	3	0.48-0.49
Chlorophyll-a	3	0.40-0.41	3	0.42	4	0.42-0.43

Table A8: List of samples removed from periphyton models due to large Cook's distance and model residual.

Variable	Samples Removed Winter	Samples Removed Summer	Samples Removed Fall
Total Abundance	2013.Winter.R2.S6.MS	2010.Summer.R3.S1.D	2012.Fall.R2.S6.MD
Total Abundance	2016.Winter.R2.S2.S	2016.Summer.R2.S6.D	2009.Fall.R2.S3.D
Total Biovolume	2013.Winter.R2.S6.MS	2010.Summer.R3.S1.D	2012.Fall.R2.S6.MD
Total Biovolume	2016.Winter.R2.S2.S	2016.Summer.R2.S6.MD	2009.Fall.R2.S3.D
Chl-a	2016.Winter.R2.S2.S	2010.Summer.R3.S1.D	2012.Fall.R2.S6.MD
Chl-a	-	2016.Summer.R2.S6.MD	2009.Fall.R2.S3.D

Appendix-A9: Summary of Benthic Invertebrate Community Composition, Abundance, Biomass Supplemental Results

Table A9: Rock Basket Recovery by Season in 2016. Fractions indicate the number of substrates recovered over the number of substrates deployed.

Season	2016
Winter	32/35
Summer	32/35
Fall	26/35

Community analyses of the 2008 – 2016 invertebrate data was also completed at the genus level. The NMDS stress index was 0.23, which indicates the two NMDS axes partially explain the invertebrate community composition. A permutational MANOVA indicated that season explained some variation (R^2=0.12) in invertebrate community compositions and was significant (F=29.7, p<0.001). The separation of invertebrate community compositions in terms of season, especially winter being different from fall and summer was visually evident.

Appendix-A10: Benthic Invertebrate Production and Fish Food Models Supplemental Results

Table A10: Summary of the number of plausible models identified using model averaging (those with a AIC <3) and the range of pseudo R2 values for selected benthic invertebrate models

Benthic Invertebrate Response	Winter		Summer		Fall	
	# of plausible models	range of pseudo R^2	# of plausible models	range of pseudo R^2	# of plausible models	range of pseudo R^2
Abundance	4	0.36-0.39	3	0.33-0.35	3	0.28-0.31
Biomass	3	0.23-0.24	4	0.10-0.11	8	0.08-0.12
Species Richness	4	0.24-0.29	4	0.17-0.18	6	0.12-0.17
Simpson's Index	2	0.31-0.33	4	0.22-0.25	4	0.44-0.45
Percent EPT	3	0.14	2	0.37	3	0.35-0.38
Percent Quality Fish Food	3	0.21	3	0.39-0.41	3	0.36-0.38

Table A11: List of samples removed from benthic invertebrate models **due to large Cook's distance and model residual.**

Variable	Samples Removed Winter	Samples Removed Summer	Samples Removed Fall
Total Abundance	2016.Winter.R2.S7.S	2009.Summer.R2.S5.M	2012.Fall.R2.S6.S
Total Abundance	-	-	2012.Fall.R2.S5.M
Total Biomass	2016.Winter.R2.S2.S	2009.Summer.R2.S5.M	2012.Fall.R2.S5.M
Total Biomass	2016.Winter.R2.S7.MS	2014.Summer.R2.S7.MS	2014.Fall.R2.S6.MS
Species Richness	2013.Winter.R2.S5.D	2009.Summer.R2.S5.M	2009.Fall.R2.S5.M
Species Richness	2013.Winter.R2.S7.D	2010.Summer.R2.S6.S	2016.Fall.R2.S2.D
Species Richness	-	2012.Summer.R2.S5.S	-
Simpson's Index	2016.Winter.R2.S3.MS	2009.Summer.R2.S5.M	2010.Fall.R2.S6.M
Simpson's Index	2016.Winter.R2.S1.D	2010.Summer.R2.S7.M	2016.Fall.R2.S6.M
Percent EPT	2014.Winter.R2.S1.D	2009.Summer.R2.S5.M	-
Percent EPT	2014.Winter.R2.S5.MS	2010.Summer.R1.S1.D	-
Percent EPT	2016.Winter.R2.S5.S	-	-
Percent Quality Forage	2016.Winter.R2.S1.S	2009.Summer.R2.S5.M	-
Percent Quality Forage	2016.Winter.R2.S5.M	2010.Summer.R1.S1.D	-

Appendix-A11: Fish Food Supplemental Results

An NMDS was run on the RBT and MWF stomach contents at the family level. The stomach contents of RBT and MWF had similar benthic invertebrate compositions, with small differences between MWF and RBT fish stomachs (R^2=0.02, F=2.52, p= 0.007) and year (R^2=0.02, F=2.33, p= 0.006), as determined by a PERMANOVA. There were a few fish caught in 2014 that had distinct community compositions (Figure A5).

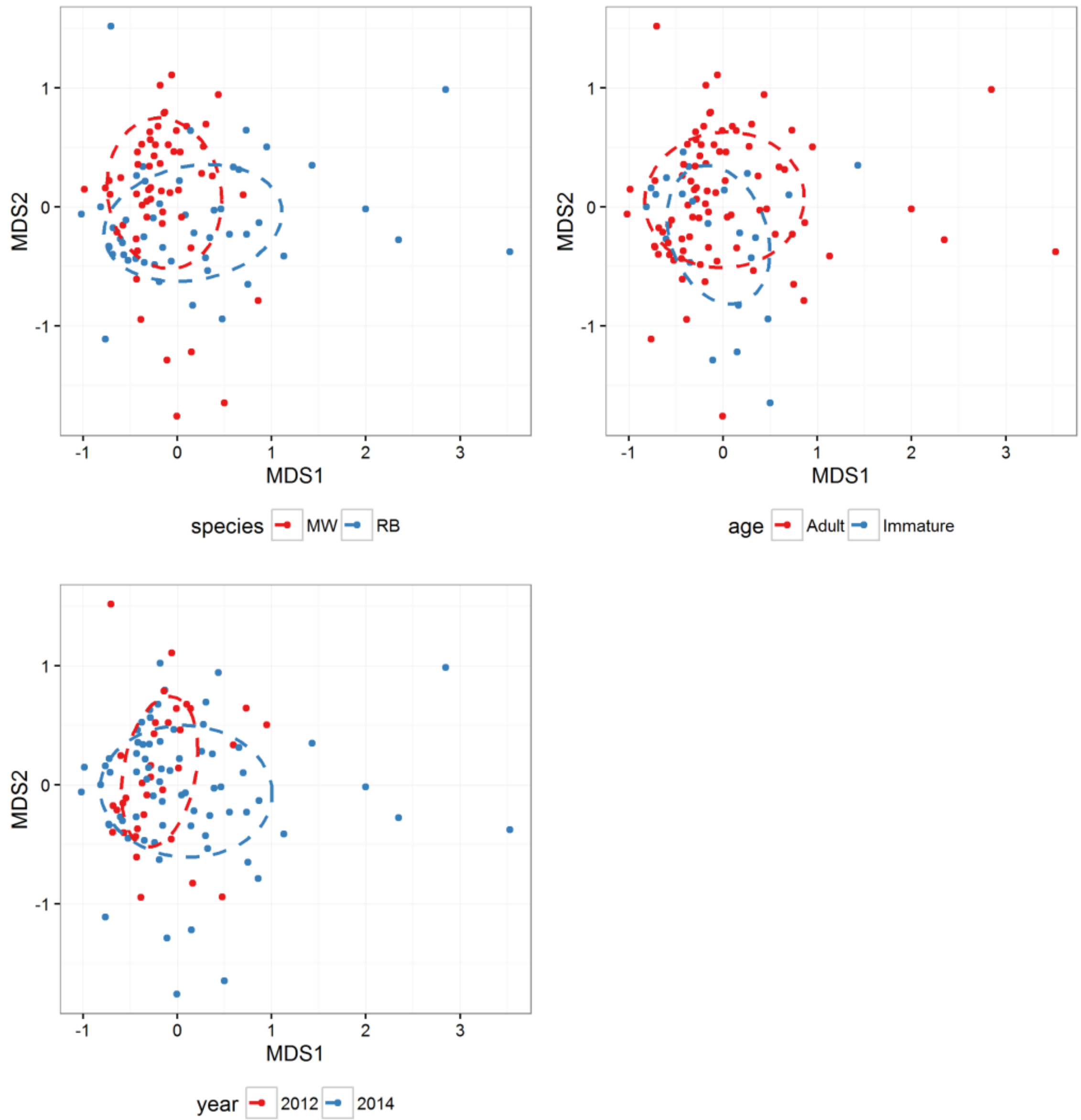

Figure A5: NMDS of benthic invertebrate community found in fish stomachs of MWF and RBT caught in Fall 2012 and 2014 grouped by species, age, and year.