Columbia River Project Water Use Plan

Lower Columbia River Fish Management Plan

Implementation Year 7

Reference: CLBMON#44

Lower Columbia River Physical Habitat and Ecological Productivity Monitoring (Year 7)

Study Period: 2014

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Lower Columbia River Physical Habitat and Ecological Productivity Monitoring (Year 7)

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ACRONYMS AND ABBREVIATIONS

µS	microsiemens
AFDW	ash free dry weight
AICc	Akaike information criterion corrected for small sample sizes
Al	aluminum
ALGS	Arrow Lakes Generating Station
ALR	Arrow Lakes Reservoir
BBK	Birchbank
BC Hydro	British Columbia Hydro and Power Authority
BRD	Combined discharge from Brilliant Dam, including spill and the Brilliant Dam expansion project
Caro Labs	Caro Environmental Laboratories (Kelowna, B.C.)
Celgar	Zellstoff Celgar Mill
CFU	colony forming unit
chl-a	Chlorophyll-a
CV	Coefficient of variation
Didymo	*Didymosphenia geminate*
DO	Dissolved oxygen
EPT	Ephemeroptera (mayflies), Plecoptera (stoneflies), Trichoptera (caddisflies)
FFF	fall fluctuating flow
FFI	Fish Food Index
HBI	Hilsenhoff Biotic Index
HLK	Hugh L. Keenleyside
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	nitrogen
n	sample size
NMDS	Non metric multidimensional scaling
NTU	nephelometric turbidity units
PCA	principal component analysis
POM	particulate organic material
RBT	Rainbow Trout
RVI	relative variable importance
SD	standard deviation
SRP	soluble reactive phosphorus
TDS	total dissolved solids
T-P	total phosphorus
TSS	total suspended solids
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
Aerobes	Organisms that require >1-2 mg/L dissolved oxygen in their environment
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
Bioaccumulation	Removal of metal from solution by organisms via adsorption, metabolism
Bioavailable	Available for use by plants or animals
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.
Inflow plume	An inflow seeks the layer of matching density in the receiving water, diffusing as it travels; High TSS, TDS and low temp increase water density
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
Macronutrient	The major constituents of cells: nitrogen, phosphorus, carbon, sulphate, H
Mainstem	The primary downstream segment of a river, as contrasted to its tributaries
Mesotrophic	A body of water with moderate nutrient concentrations
Micronutrient	Small amounts are required for growth; Si, Mn, Fe, Co, Zn, Cu, Mo etc.
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
Myxotrophic	Organisms that can be photosynthetic or can absorb organic materials directly from the environment as needed
Nano plankton	Minute algae that are less than 5 microns in their largest dimension
Pico plankton	Minute algae that are less than 2 microns in their largest dimension
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
Redox	The reduction (-ve) or oxidation (+ve) potential of a solution
Reducing envi	Devoid of oxygen with reducing conditions (-ve redox) e.g. organic sediments
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 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 1-1 summarizes the management questions, hypotheses and results to date.

LCR flows came from the Hugh L. Keenleyside Dam (52.3%), and from the Kootenay River (45%) in 2014. Freshet flows were on par with typical years and the peak flow occurred in early July (3,677.9 m^3/s on July 8th at Birchbank gauging station). Regression modeling 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 during pre-MWF flows than 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, ranging from approximately 3 to 19°C in 2014. Regression modeling during each of the flow periods of cumulative data to date indicated that the influence of flow on water temperature was relatively weak compared to other model predictors such as air temperature, reservoir temperature and reservoir elevation.

A suite of water quality parameters were collected on four occasions in 2014 and indicated good water quality in both the Kootenay and LCR. This was the first year that modelling was used to statistically explore if flow management alters the availability of biological active nutrients and/or the electrochemistry of LCR. The results are preliminary due to data limitations that result from infrequent annual/seasonal sampling. Modelling showed that variability in flow was a key predictor of LCR nutrients. Modelling did not reveal key predictors of electrochemistry. Further analysis is needed to understand the influence of managed flow periods (MWF and RBT) on water quality. It is suspected that managed flows have a subtle effect compared to the overwhelming effects of freshet. We anticipate that the managed flows can cause small decreases in electrochemistry parameters through dilution, and may improve particulate and dissolved nutrient delivery under low to moderate flow conditions, but fish flows are unlikely to have a discernible effect on pH, dissolved oxygen concentrations, or on the overall nutrient status of the LCR.

Numerous benthic productivity metrics were sampled during the winter, summer and fall using artificial substrate samplers that were deployed on the river bottom for 10 weeks. The sampling revealed periphyton and benthic invertebrate communities that were productive, diverse and variable. Most production metrics were comparable to those from other large, moderately productive rivers. Modeling demonstrated that key factors controlling periphyton and invertebrate production shifted seasonally and included velocity, flow variability, substrate size and to a lesser extent nutrient loading in Arrow Lakes Reservoir. The results suggest that a direct link between productivity and operations may exist. Production modelling results are also considered preliminary and further analysis with additional years of data is needed to better understand how flow variability and operations may affect benthic productivity.

Three 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), % biomass of Chironomidae and good quality forage (percent biomass of EPT + Diptera). The availability of food for fish was greatest in areas with larger substrates and higher velocities.

Table 1-1: CLBMON-44 Status of Objectives, Management Questions and Hypotheses After Year 7

Management Questions	Management Hypotheses	Year 7 (2014) 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.	Regression modeling of the studies cumulative data to date indicates that the influence of flow on LCR water temperature is relatively low compared to other model predictors. When all flow periods were considered, LCR water temperatures were most strongly correlated with air temperature and reservoir water temperature. Flow was positively associated with river temperature during the MWF and FFF periods, and negatively associated with river temperature during the RBT flow period. Based on this analysis, flow is not an important determinant of river temperature. These findings are consistent with that reported by Scofield *et al.* (2011) and Olson-Russello (2014) for previous years of the study. Given the nominal 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.	Regression modeling suggests 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 (2008-2014) flow periods. We therefore reject all three null hypotheses.
	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.	
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.	Water quality parameters that address electrochemistry include: conductivity, TDS, hardness, alkalinity, dissolved metals ions and pH. Biologically active nutrient parameters include: nitrate, ammonia, total P and ortho phosphate (SRP). Based on data collected throughout the study, LCR has good water quality. Parameters rarely exceeded water quality guidelines or objectives. Due to the limited water quality sampling regime (3-4 collections per year) it has been difficult to statistically test whether flows within each flow period have an effect on water quality. Variability in flow had a positive effect on the availability of nutrients (No^2+No^3 and total
	Ho3Aphy: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the electrochemistry 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.	phosphorus). Operations during the MWF and RBT flow periods were also factors in predicting total phosphorus, but were less important than variability in flow. Modelling of electrochemistry parameters was not informative. Although these initial results are consistent with what has been previously reported, additional modelling is necessary to further understand what is driving the water quality in LCR. Based on our understanding of the system to date, we believe that the influence of fish flows on water quality is subtle compared to the stronger effects on water quality in freshet, anthropogenic nutrient donation, groundwater inputs, and even photosynthesis within LCR. We anticipate that fish flows may cause small decreases in electrochemistry parameters through dilution, and may improve particulate and dissolved nutrient delivery under low to moderate flow conditions, but that they are unlikely to have a discernible effect on pH, or on the overall nutrient status of LCR. We therefore continue to tentatively accept the management hypotheses $HO_{3phy,}$ $HO_{3Aphy,}$ and HO_{3Bphy} and assume that fish flows, whether they be MWF, RBT or FF flows, have no effect on the water quality 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.	Regression modelling indicated that velocity is an important determinant of the benthic invertebrate community. Variability in flow was also important during the MWF flow period, and to a lesser extent during RBT and FFF periods. These modelling results suggest that there may be a direct link between operations and benthic invertebrate production. The results are preliminary as additional analysis is needed to further elucidate relationships and to understand how flow variability and operations affect the benthic invertebrate community. At this time, we continue to tentatively reject all four null hypotheses.
	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.	Similar to benthic invertebrates, when considering all flow periods and metrics, regression modelling indicated that velocity was the most important determinant of the periphyton community. Variability in flow was also important. This result suggests that a direct link between productivity and operations may exist. Since this is the first attempt to explicitly test the management questions through modelling, results are considered preliminary and further analysis with additional years of data is needed to better understand how flow variability and operations may
	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.	affect periphyton productivity. We tentatively reject Ho2 A B and Ceco, that RBT, FFF and MWF flows 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	Regression modelling indicated that velocity and substrate score were important determinants of the benthic invertebrate community that is considered high quality forage by fish. Although there was some variation by flow period, high quality forage was positively associated with velocity and substrate size. We continue to tentatively reject all four null hypotheses because operational changes have a downstream effect on velocity and ultimately the availability of food for fish. These effects are relevant across all flow periods.
	Ho_{3Aeco}: Continued implementation of MWF flows does not increase availability of fish food organisms in LCR.	
	Ho_{3Beco}: 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.	

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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 salmonid 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 7 (2014) of the study and includes both historic and 2014 data pertaining to the hydrology, water quality 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

1) What are the composition, abundance, and biomass of epilithic algae and benthic invertebrates in LCR?
2) 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?

3) 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 quality, 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**). Periphyton and macroinvertebrate productivity monitoring took place at seven different productivity monitoring sites within reach 2 during three different seasons (**Figure 2-2 and Table 2-1**).

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

Station Name	Sample Type	UTM Coordinates	
		Northing	Easting
WQIS1	Physical/chemical/water level	5,465,742	445,693
WQIS2	Physical/chemical/water level	5,464,573	450,072
WQIS3	Physical/chemical/water level	5,464,517	452,244
WQIS4	Physical/chemical/water level	5,455,332	452,653
WQIS5	Physical/chemical/water level	5,450,221	448,514
WQ C1 (Norns Creek)	Physical/chemical	5,465,356	451,746
WQ C2 (Kootenay)	Physical/chemical/water level	5,462,911	454,114
R2-S1	Periphyton and macroinvertebrate substrates / temp / light	5,464,323	451,486
R2-S2	Periphyton and macroinvertebrate substrates / temp / light	5,464,428	451,942
R2-S3	Periphyton and macroinvertebrate substrates / temp / light	5,463,822	452,971
R2-S4	Periphyton and macroinvertebrate substrates / temp / light	5,463,186	452,592
R2-S5	Periphyton and macroinvertebrate substrates / temp / light	5,463,085	452,789
R2-S6	Periphyton and macroinvertebrate substrates / temp / light	5,464,256	452,488
R2-S7	Periphyton and macroinvertebrate substrates / temp / light	5,463,032	452,480

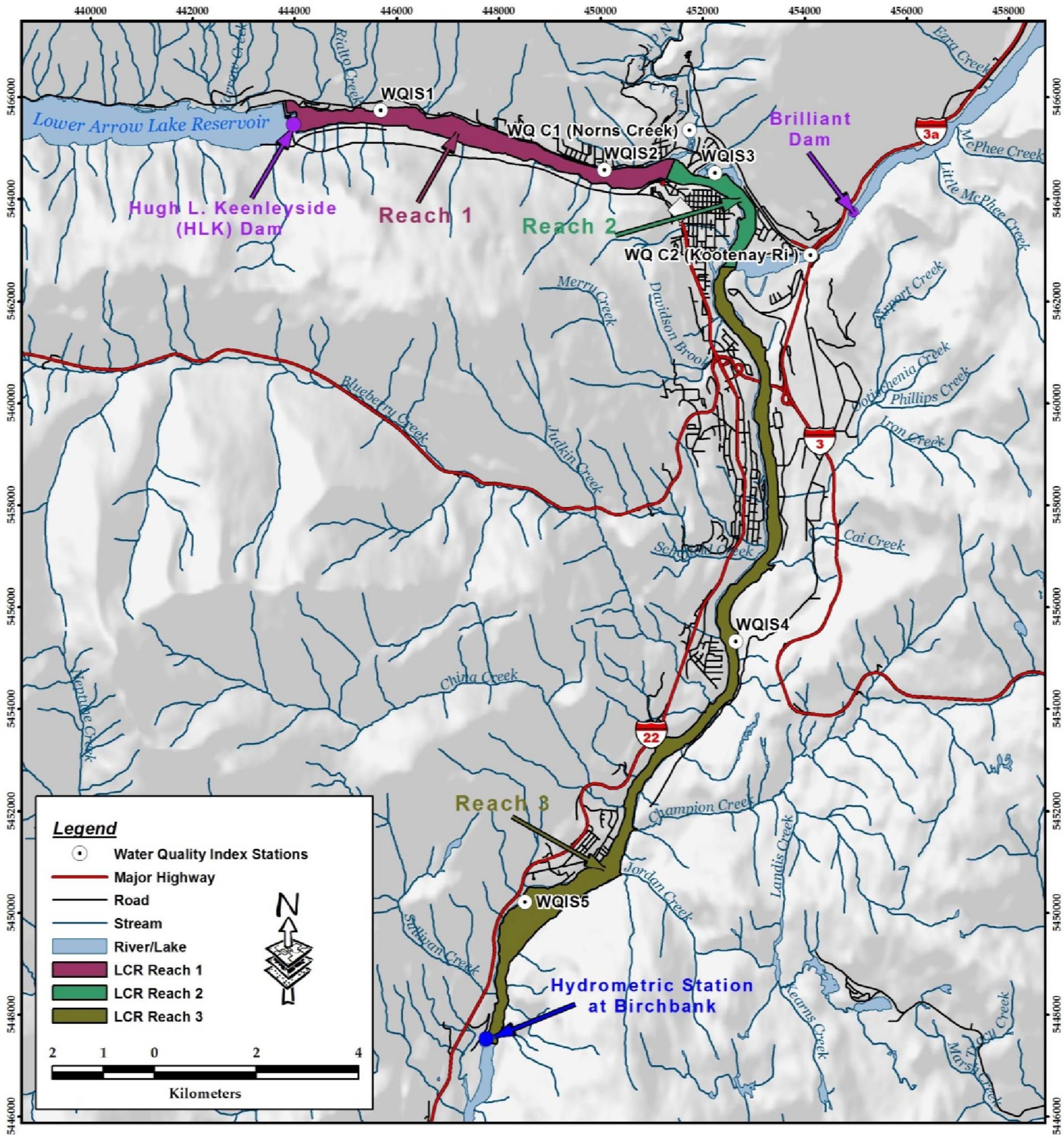


Figure 2-1: Map of Lower Columbia River Study Area and Water Quality Index Station Sampling Locations

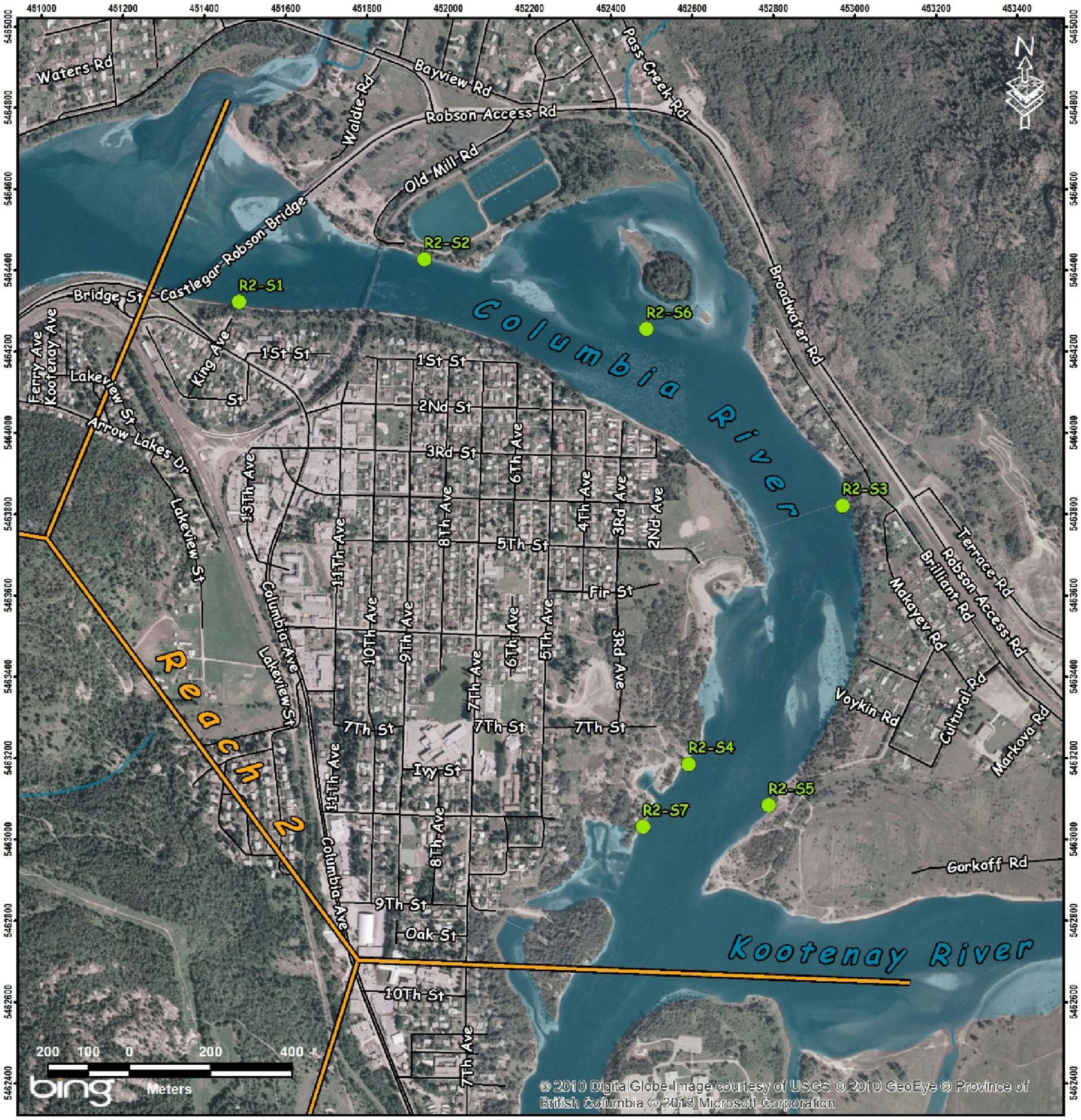


Figure 2-2: Benthic Productivity Sampling Locations in 2014.

2.2 Hydrology and Water Level

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**).

River flow and discharge data were obtained from Robyn Irvine of Poisson Consulting Ltd. 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 for all of 2014, and specific comparisons of the three different flow periods were undertaken.

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]. Previously used level loggers were available as backup, and therefore, 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 2013, it was discovered that two of the installed sensors had failed and were no longer collecting accurate elevation and temperature data. The sensors were removed and sent back to the manufacturer for repair. The repaired sensors were replaced in June 2014.

[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

Chemical and physical water quality parameters were collected at seven different sampling locations during 2014 (**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).

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).

Field trips were conducted on March 27, June 4, August 14, and October 23 during 2014, with all sampling occurring during day-time hours. The following field water quality parameters: temperature, dissolved oxygen (DO), percent dissolved oxygen saturation, pH, conductivity, and total dissolved solids (TDS) were measured with a pre-calibrated Hannah HI 9828 sonde, by lowering the probe 1 m below the water's surface. Readings were simultaneously recorded in the multi-meter memory and in a field book.

Water quality samples were collected in a low-metals bottle Van Dorn sampler. They were collected from the mid-water column (2-8 m depth) or 1 m below the surface if flows were too high to use the bottle sampler. Water depths were measured with a Lowrance depth sounder. Every mainstem LCR sample was a composite of three subsamples collected from: one third of the river width from left bank, mid river and one third of the river width from right bank. These subsamples were mixed in a triple-rinsed 4L container before decanting into the sample bottles. A composite sample of the river transect was collected because the focus of the sampling effort is to understand the water quality of the river as a whole versus the water quality from the sample points mentioned above.

The sample bottles were provided by Caro Environmental Laboratories (Caro Labs) with the appropriate preservatives pre-measured into the bottles. The non-filtered samples were analyzed for total hardness, ammonia as nitrogen (N), nitrate as N, nitrite as N, total phosphorus, ortho-phosphorus, TDS, total suspended solids (TSS) and turbidity according to Standard Methods. Field-filtered samples were analyzed for low-level soluble reactive phosphorus (SRP) and total dissolved solids (TDS). The filled sample bottles were placed on chipped ice and delivered to Caro Labs in Kelowna, BC within 24 hours of collection. One randomly chosen field duplicate and one deionized water travel blank were collected on each field trip. Additional QA/QC protocols were undertaken at Caro Labs.

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. Productivity sampling in Years 5 and 7 differed from Years 1-3, in that all sampling locations were located in Reach 2 and were sampled during summer, fall and winter. In addition, the depths sampled at each site were increased from three 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 2-2**).

Table 2-2: 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

A single artificial sampler apparatus was used for all seasons in 2014 (**Figure 2-3**). The apparatus was consistent with samplers deployed during the first winter deployment in 2013 (Larratt *et al.* 2013). During the last sampling year (2012-2013) two different apparatus designs were used in order to accommodate SARA permit #245 under Section 73 of the Species at Risk Act. That permit allowed for shorthead sculpin (*Cottus confusus*) to be incidentally collected, then released unharmed at the site of capture annually after August 15th. The shorthead sculpin was since downgraded in March 2013 and is now listed as Special Concern, and therefore a SARA permit is no longer required.

In 2014, all sampling sessions were 10 weeks in duration. The winter samplers were deployed from January 15th through March 27th. The sampling session was designed to coincide with the MWF flow period. The summer sampling period occurred from June 5th through August 13th and the fall sampling period occurred from August 15th through October 23rd. The winter and fall sampling sessions entirely overlap with MWF and FF flows, while only the first month of the summer deployment overlaps with the RBT flow period. **Table 2-3** provides deployment dates, sampling numbers and recovery rates.

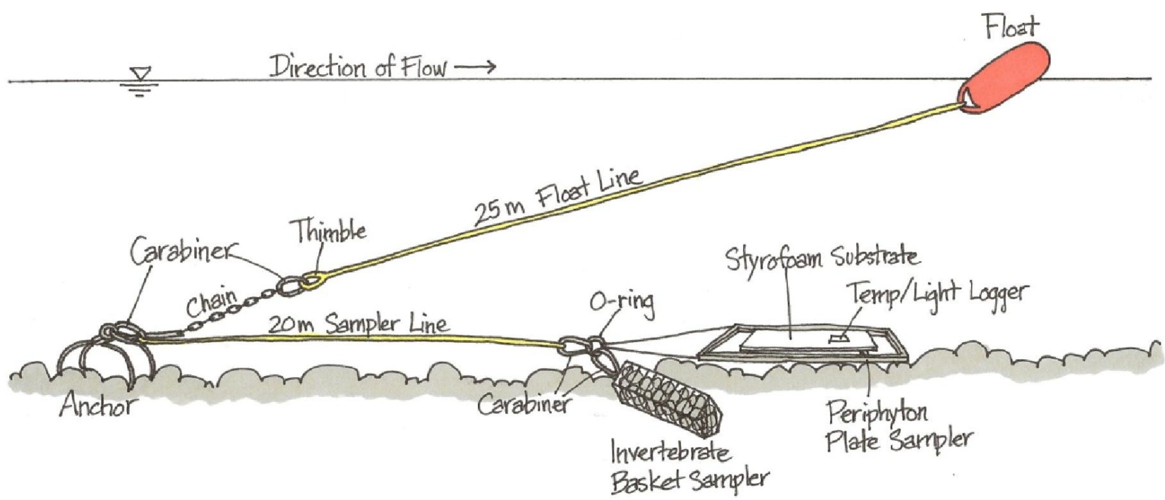


Figure 2-3: Diagram of the Periphyton and Macroinvertebrate Sampling Apparatus Deployed in Winter, Summer and Fall (2014)

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

Table 2-3: Artificial Sampler Deployment and Recovery Rates in 2014

Season	Reach	Site	Periphyton Samplers		Invertebrate Basket Samplers	
			# Deployed	# Retrieved (% Recovery)	# Deployed	# Retrieved (% Recovery)
Winter (Jan 15 - Mar 27) 10 weeks	2	Site 1 (S1)	5	4 (80)	5	4 (80)
		Site 2 (S2)	5	5 (100)	5	5 (100)
		Site 3 (S3)	5	5 (100)	5	5 (100)
		Site 4 (S4)	5	5 (100)	5	5 (100)
		Site 5 (S5)	5	4 (80)	5	3 (60)
		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 5 - Aug 13) 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	4 (80)	5	4 (80)
		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	5 (100)
Summer Totals			**35**	**32 (91)**	**35**	**33 (94)**
Fall (Aug 15 - Oct 23) 10 weeks	2	Site 1 (S1)	5	5 (100)	5	5 (100)
		Site 2 (S2)	5	4 (80)	5	4 (80)
		Site 3 (S3)	5	4 (80)	5	4 (80)
		Site 4 (S4)	5	4 (80)	5	4 (80)
		Site 5 (S5)	5	4 (80)	5	3 (60)
		Site 6 (S6)	5	5 (100)	5	5 (100)
		Site 7 (S7)	5	5 (100)	5	5 (100)
Fall Totals			**35**	**31 (88)**	**35**	**30 (85)**
2014 Totals			**105**	**96 (91)**	**105**	**95 (90)**

2.4.1.2 Winter Accrual Data Collection

This was the first year that accrual sampling, designed to investigate periphyton biomass accrual rates and test management hypothesis Ho_{2eco}, was completed during the winter sampling period. 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, a random number generator was used to take four Styrofoam punches from each sampler to assess the following metrics: 1) chl-a-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 in future years. The contents from each bucket were then captured on a 100µm 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 completed by H. Larratt, with QA/QC and taxonomic verifications provided by Dr. Stockner. Fresh, chilled samples were examined within 48-hours for protozoa and other microflora that cannot be reliably identified from preserved samples. One punch was preserved using Lugol's solution and was stored until taxonomic identification and biovolume measurements could be undertaken. Species cell density and total biovolume were recorded for each sample. A photograph archive was compiled from LCR samples. Detailed protocols on periphyton laboratory processing are available from Larratt Aquatic.

Periphyton datasets from 2014 and previous years of the study (2008 – 2010, 2012) 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 macro invertebrates 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 Statistics Procedures

All statistical analyses and the creation of most figures were conducted in R (R Development Core Team 2013). Prior to carrying out statistical analyses, 2014 data was combined with datasets from previous years (2008-2013).

2.5.1 Water Levels

The mean 2014 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 2014 analysis differed from previous years, in that the whole annual dataset was used to predict elevations, rather than only a subset of the flow period. This change in methodology was undertaken to improve 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 (2008-2014). Differences among predicted elevations during each time 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 - 2014 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 (2008-2014). 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 2008-2014 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 WQI5 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:

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

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:

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

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 on 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 Water Quality

Water quality data (2014) was combined with datasets from previous years (2008-2013). Data consisted of point samples from each WQIS collected four times annually and analyzed for approximately 15 parameters. Consistent with previous years, if a measurement was non-detectable, it was entered into the database as ½ the lab reportable detection limit.

To illustrate the variation between season and year, boxplots of eight water quality parameters were generated using R. Sample number (n) at each LCR site and season ranged from 2 to 17. Winter had the smallest sample number (n=2), spring and fall were intermediate (n=7 and 8) and summer had the greatest (n=17).

The hypothesis Ho_{3phy}, states that the continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall does not alter the electrochemistry and biologically active nutrients of LCR. To explore the relationship between flow and LCR water quality, explanatory variables were derived using flow from HLK, Arrow Lakes Reservoir (ALR) water chemistry data and ALR nutrient addition data. The LCR water quality responses of interest included: conductivity, total dissolved solids, total phosphorus as phosphorus and nitrate + nitrite. LCR water quality data was available from 2008 to 2014, with most data collected during the spring (Apr 1 – Jun 30), summer (Jul 1 – Sept 30) and fall (Oct 1 – Dec 31). Because of this, winter was excluded from this analysis.

Table 2-5: Predictor Variables for the Water Quality Linear Regression Analysis

Predictors	Description
Year	Years included 2008 - 2014
Season	Spring date range: Apr 1 – Jun 30, Summer date range: Jul 1 – Sept 30, Fall date range: Oct 1 – Dec 31.
Flow Daily SD	The mean of the standard deviation of daily flow from HLK. This predictor describes the average daily variation of flow.
Alkalinity	Alkalinity measured at station AR8 in ALR. Alkalinity measurements collected within a season were averaged.
Dissolved Nitrate	Nitrate measured at station AR8 in ALR. Dissolved nitrate measurements collected within a season were averaged.
Total Nitrogen	Nitrogen measured at station AR8 in ALR. Total nitrogen measurements collected within a season were averaged.
Total Phosphorus	Total phosphorus measured at station AR8 in ALR. Total phosphorus measurements collected within a season were averaged.
Total Nutrition	Nutrients added to ALR through the Arrow Lakes Reservoir Nutrient Restoration Program. Calculated as the sum of nutrient addition, in metric tons, over each season.
Elevation diff. (MWF)	Calculated for each season, it is equal to the maximum elevation of the first half of the season minus the minimum of the second half of the season. It is referred to as MWF because the methodology originated from the water elevation analysis for MWF.
Elevation diff. (RBT)	All elevation drops within each season were summed. It is referred to as RBT because the methodology originated from the water elevation analysis for RBT that described the drop in elevation during the RBT flow period.

We used linear mixed-effects modeling (Zuur *et al.* 2008) and AICc model selection to evaluate the relative effects of the predictors on each response. Predictor variables for the water quality linear regression analysis are described in Table 2-5. 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. Variables having RVI values above 0.55 and confidence intervals that did not span zero were considered the most relevant.

2.5.4 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 2008-2014 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 to year, season, depth and transect. The amount of variability in community composition explained by year, season, depth, and transect 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 2.0-10 (Oksanen *et al.* 2013).

2.5.5 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 2-6 provides a description of the explanatory variables used for both periphyton and benthic invertebrates.

Table 2-6: Explanatory Variables for both Periphyton and Benthic Invertebrates

Variable	Description
Added Nutrients	Nutrient loading into Arrow Lakes Reservoir through the Arrow Lakes Reservoir Nutrient Restoration Program. This explanatory variable describes the total nutrients added four months prior to sampling. No nutrient addition occurred prior to the winter deployment.
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 period only. In addition to Flow Daily SD, the standard deviation of flow across the deployment period and the coefficient of variance were also considered. Each had similar results, so only Flow Daily SD was incorporated into the models.
Elev. Diff. (MWF)	Calculated for each deployment, it is equal to the maximum elevation of the first half of the deployment minus the minimum of the second half of the deployment. It is referred to as (MWF) because the methodology originated from the water elevation analysis for MWF. Even though it was calculated for each deployment period, it was only used in winter models.
Elev. Diff. (RBT)	All elevation drops within each season were summed. It is referred to as (RBT) because the methodology originated from the water elevation analysis for RBT that described the drop in elevation during the RBT flow period. Similar to Elev. Diff. (MWF) this was calculated for all seasons, yet was only used in the summer model.

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 **Tables 2-7** and **2-8**. Upon inspection of the residual plots for periphyton: total abundance, total biovolume, percent community from reservoir, and percent good forage were log transformed to reduce heteroscedasticity.

Table 2-7: Responses for Periphyton

Variable	Description
Total Abundance	Total Abundance across all species
Total Biovolume	Total Biovolume across all species
Chl-a	Total Chlorophyll-a
Species Richness	Number of different species found
Simpson's Index	A measure of diversity which takes into account the number of species present, as well as the relative abundance of each species
Percent from Reservoir	The percentage of the periphyton community collected that originated in the reservoir. This was calculated by taking the biovolume of plankton and dividing by the total biovolume
Percent Good Forage	The percentage of the biovolume that is classified as good forage for fish.

The following datasets for benthic invertebrates were log transformed: total abundance, total biovolume, percent chironomidae, percent EPT, and percent quality forage (**Table 2-8**).

We used linear mixed-effects modeling (Zuur *et al.* 2008) 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.

Table 2-8: 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
Hilsenhoff Biotic Index	The index incorporates the sensitivity and abundance of different taxonomic groups to low oxygen conditions. In this case, the HBI index is useful because it may detect community shifts from taxa such as Chironomidae or Oligochaeta to Ephemeroptera / Plecoptera / Trichoptera as flows increase within side channel areas. The Hilsenhoff Biotic Index is calculated as follows: $$HBI = \sum \frac{x_i t_i}{n}$$ where xi is the number of individuals within a taxon, ti is the tolerance value of the taxon (from published literature), and n is the total number of organisms in the sample (Plafkin *et al.* 1989).
Percent Chironomidae	Calculated by taking the biomass of Chironomidae and dividing by the total biomass.
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.

2.5.6 Fish Food

Three 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), % biomass of Chironomidae and good quality forage (percent biomass of EPT + Diptera). A single response variable for periphyton (percent good forage) was designed to test the availability of food for fish.

3.0 RESULTS

3.1 Hydrology

3.1.1 River Flows

Flow within the study area is dominated by discharges from HLK Dam on the Columbia River and the Brilliant Dam on the Kootenay River. The sum of these flows and of other smaller, local tributaries is recorded at the Birchbank gauging station. In 2014, the mean daily river flows from the Columbia and Kootenay Rivers were 52.3% and 45%, respectively, of the total flows at the Birchbank gauging station. This constituted 97.3% of the total flow, with the remaining 2.7% originating from smaller tributaries such as Norns Creek and outfalls.

Figure 3-1 depicts the 2014 hydrographs of mean daily river flows from LCR at HLK Dam, Kootenay River from the Brilliant Dam and at the Birchbank gauging station. The mean daily river flows at HLK Dam were greater than those at Brilliant Dam (1089.1 m^3/s and

936.1 m³/s, respectively), but Brilliant exhibited a higher peak, with a maximum flow of 2535.9 m³/s recorded on May 26th (**Table 3-1**).

The highest flow recorded at the Birchbank gauging station in 2014 was 3,677.9 m³/s on July 8th. This is compared to 2011, 2012 and 2013 when the peak was 4,155.4 m³/s on July 9th, 6,043.1 m³/s on July 21st and 4,434.4 m³/s on July 5th, respectively (Olson-Russello *et al.* 2014, Larratt *et al.* 2013; Olson-Russello *et al.* 2012).

Table 3-1: Mean Daily River Flows (m³/s) at HLK Dam, Brilliant Dam and the Birchbank Gauging Station in 2014

Location	N (days)	Statistic	2014
HLK	365	Mean	1089.1
		Min	500.6
		Max	2264.2
		SD	466.2
Brilliant	365	Mean	936.1
		Min	401.0
		Max	2535.9
		SD	580.6
Birchbank	365	Mean	2081.6
		Min	1022.4
		Max	3677.9
		SD	712.7

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 A-1)**. During the MWF flow period (Jan 1 – Mar 31), flows at HLK Dam, Brilliant Dam and the Birchbank gauging station exhibited a different flow pattern compared to earlier years of the study. Prior to 2013, flows originating from HLK and Brilliant Dams during the MWF flow period were fairly consistent from the beginning to the end of the flow period. In 2013, the flows from HLK Dam remained high (~2000 m³/s) throughout January and then showed a substantial drop on February 9th to approximately 800 m³/s. This was an approximately 60% drop in flow over a single day (Olson-Russello *et al.* 2014). In 2014 the flow started off around 1500 m³/s and gradually tapered downward throughout the flow period to approximately 750 m³/s. This was a drop of approximately 50% in flow over the 90 day flow period (**Figure 3-1**). Flows from Brilliant Dam were similar to previous years, with consistent flows throughout the MWF flow period that hovered around 500 m³/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 is not known whether the drop in flows from HLK was substantial enough to expose MWF eggs.

During the RBT flow period (Apr 1 – Jun 30), flows at Brilliant Dam steadily increased from April 1 through the end of May where they peaked at over 2,500 m³/s. The flows then declined over the remainder of the flow period until they reached about 1700 m³/s. The flows at the HLK Dam were generally held stable throughout the RBT flow period and did not substantially increase until the end of the flow period on June 28. This flow pattern was generally consistent with previous years.

During the fall fluctuating flow period, a downward trend of mean daily flow for HLK was observed. Flows from Brilliant Dam were minimal and steady throughout the flow period at approximately 400 m^3/s.

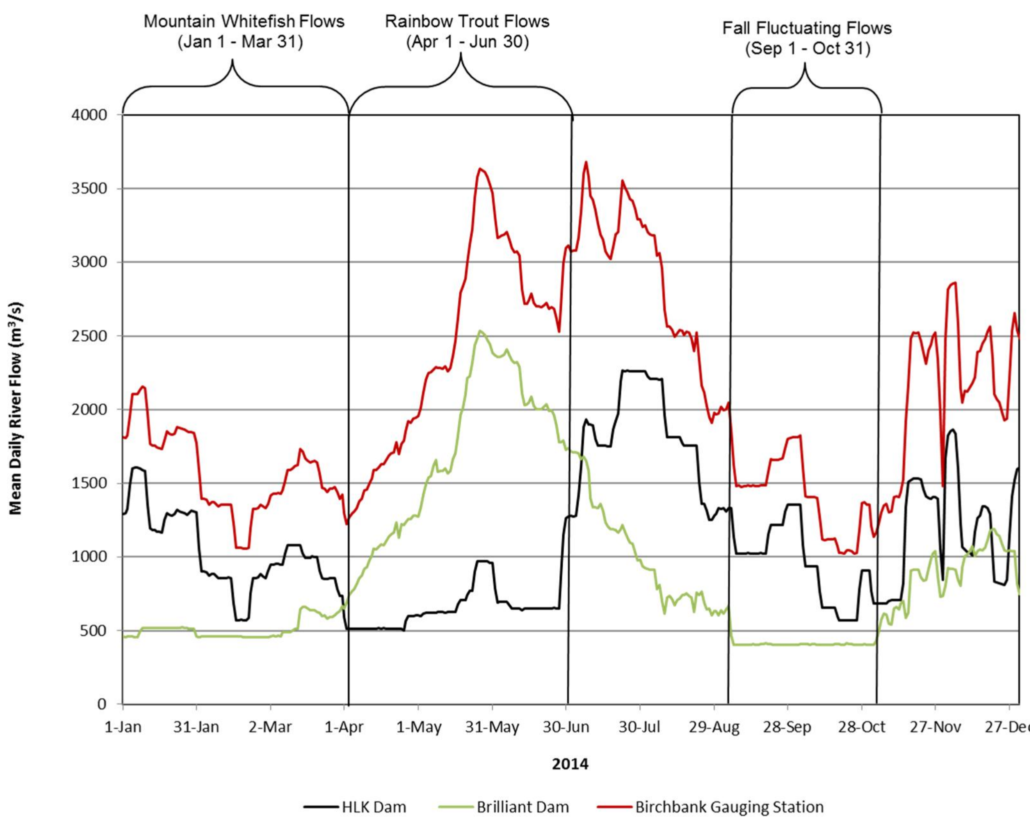


Figure 3-1: Mean Daily River Flow at HLK Dam (Columbia River), Brilliant Dam (Kootenay River), and Birchbank Gauging Station in 2014

3.1.2 Water Levels

Water level sensors collected data at all six sites throughout 2014. At WQIS1, the pressure sensor had malfunctioned and was sent to the manufacturer for repair. The repaired sensor was not installed until June 2014 and therefore WQIS1 data is only displayed from June through October (**Figure 3-2**). Data collection at the Kootenay River site (WQ C2) is displayed between January and August, as it was not possible to download the data at either the August or October field visits due to low flows and our inability to access the top of the gage (**Figure 3-2**). There is also very short periods of missing data at WQIS4 and 5 during the latter half of February when sensors at these sites were briefly exposed to air. Sensors continue to log and data will be downloaded approximately four times in 2015.

In 2014, recorded water level elevations above the Kootenay River confluence ranged from approximately 417.7 to 422 m asl. Below the confluence (WQIS4 and 5), elevations ranged from approximately 411 to 417 m asl. Index stations 4 and 5 exhibited a higher variation when compared to WQIS1-3, likely due to the combined influence of flows from both HLK and BRD dams.

Upon first glance at **Figure 3-2,** the mean daily water levels recorded at the six water quality index stations in 2014 appeared high compared to other years. At stations 1-4, the 2014 elevations were substantially higher than the mean water levels throughout the duration of the study. Unfortunately, this data is misleading because elevation data at each of these stations was lost in 2012 during record high flows. The only station that successfully captured 2012 data was WQIS5. This graph shows 2014 data that is much more aligned with the mean daily water level recorded throughout the duration of the study, and is also within the range of the calculated SD. This suggests that water levels recorded in 2014 were average and that it was not a record high flow year. It may be possible to use a GAM model, or other analytical techniques, to estimate the missing 2012 data, and to improve the accuracy of the SD. This has not yet been undertaken, but may be considered in future years.

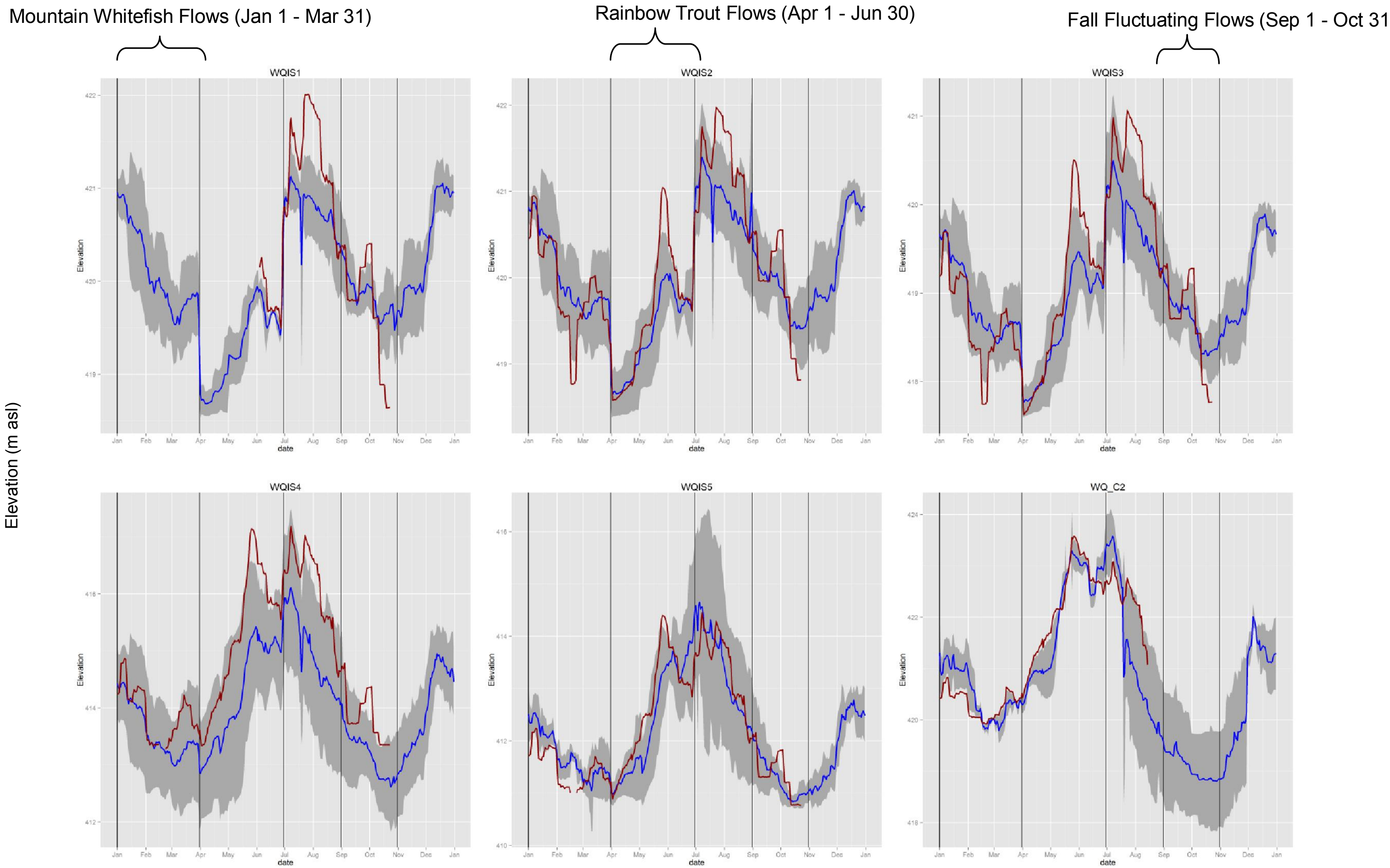


Figure 3-2: Mean Daily Water Levels Recorded at WQIS1 – 5 on LCR and at WQ C2 on Kootenay River. The red line depicts the mean daily water level recorded at each site in 2014. The blue line is the mean daily water level throughout the duration of the study (2008-14) ± SD (gray shaded area). The SD is 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

The following results address sub-hypothesis HO_{2Aphy}, which states that 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). All relationships between flow and elevation were statistically significant ($p < 0.05$). The best models varied among the five WQIS sites and contained different sets of explanatory flow variables. For all WQIS, 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, $p<0.001$) (**Figure 3-3**). 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 3-2**). 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 (**Figure 3-3**).

These results suggest 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. These results are consistent with findings by Scofield *et al.* (2011).

Table 3-2: 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(-0.000419 ± 6.919e-05) + BRD^2 (3.27e-07 ± 3.115e-08) + HLK(0.0026± 7.628e-05) + HLK^2(-3.421e-07 ± 2.997e-08)	0.899	< 2.2e-16
WQIS2	417.09 + BIR(3.807e-04 ± 5.355e-05) + BIR^2(-2.362e-08 ± 1.010e-0) + BRD(-4.586e-04 ± 4.707e-05) + BRD^2(2.780e-07 ± 1.817e-08) + HLK(2.891e-03 ± 6.452e-05) + HLK^2(-5.229e-07 ± 2.505e-08)	0.959	< 2.2e-16
WQIS3	416.48 + BIR(6.297e-0 ± 4.235e-05) + BIR^2(-3.340e-08 ± 7.989e-09) + BRD(-5.129e-04 ± 3.723e-05) + BRD^2(3.277e-07 ± 1.437e-08) + HLK(1.710e-03 ± 5.102e-05) + HLK^2(-2.116e-07 ± 1.981e-08)	0.97	< 2.2e-16
WQIS4	409.4+ BIR(3.260e-03 ± 1.195e-04) + BIR^2(-2.018e-07 ± 1.751e-08) + BRD(-4.427e-04 ± 8.091e-05) + HLK(-4.139e-04 ± 8.109e-05)	0.908	< 2.2e-16
WQIS5	409.1+ BIR(1.672e-03 ± 8.009e-05) + BIR^2(-7.976e-08 ± 8.954e-09) + BRD(-1.418e-04 ± 7.906e-05) + BRD^2(1.149e-07 ± 1.808e-08) + HLK(-3.448e-04 ± 8.567e-05) + HLK^2(1.910e-07 ± 2.634e-08)	0.98	< 2.2e-16

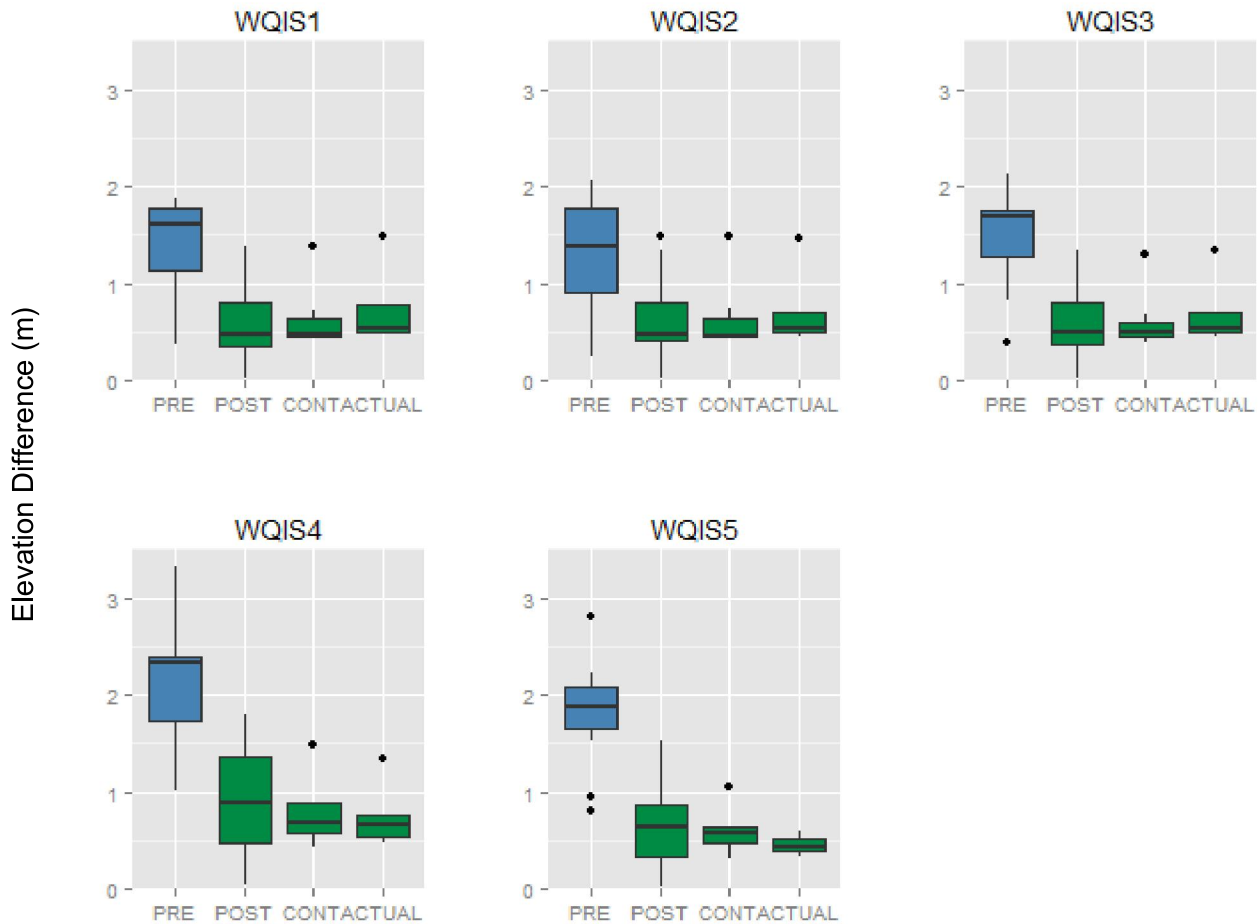


Figure 3-3: Predicted Water Level Elevation Difference between Maximum Flows during Mountain Whitefish (MWF) spawning (Jan 1 – Jan 21) and Minimum Flows during MWF Egg Incubation (Jan 22 – Mar 31) for Pre (1984 – 1994), Post (1995-2007), and Continuous (2008-2014) Flow Years at each Water Quality Index Station. Different colours indicate statistical significance ($p<0.05$) as determined by a permutation ANOVA. The “actual” dataset is included to illustrate variability between the predicted CONT values and actual elevation field data collected during 2008-2014.

3.1.2.2 Rainbow Trout Flow Period

The following results address sub-hypothesis HO_{2Bphy}, which states that continued implementation of RBT flows does not maintain constant water level elevations at Norn's Creek fan between April 1 and June 30 and are derived from analyses described in the previous section. The best models for the two sites (WQIS1 and WQIS2) that occur in close proximity to Norn's Creek fan included BBK, BRD and HLK flows (**Table 3-2**). In both cases, flow had a strong positive effect on water elevation. It is not fully understood why BRD flows have such a strong effect, given that the sites are upstream of the Kootenay River confluence. However, since our primary objective was to describe elevation as accurately as possible, these explanatory variables were left in the analysis, despite possible concerns of using co-linear explanatory variables.

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 (2008-2014) flow periods (permutation ANOVA, d.f. 3, $p<0.001$, **Figure 3-4**). Similar to the results for MWF, field measured elevations were similar to the predicted elevations. The data suggests there is a reasonable confidence in predicted versus observed values.

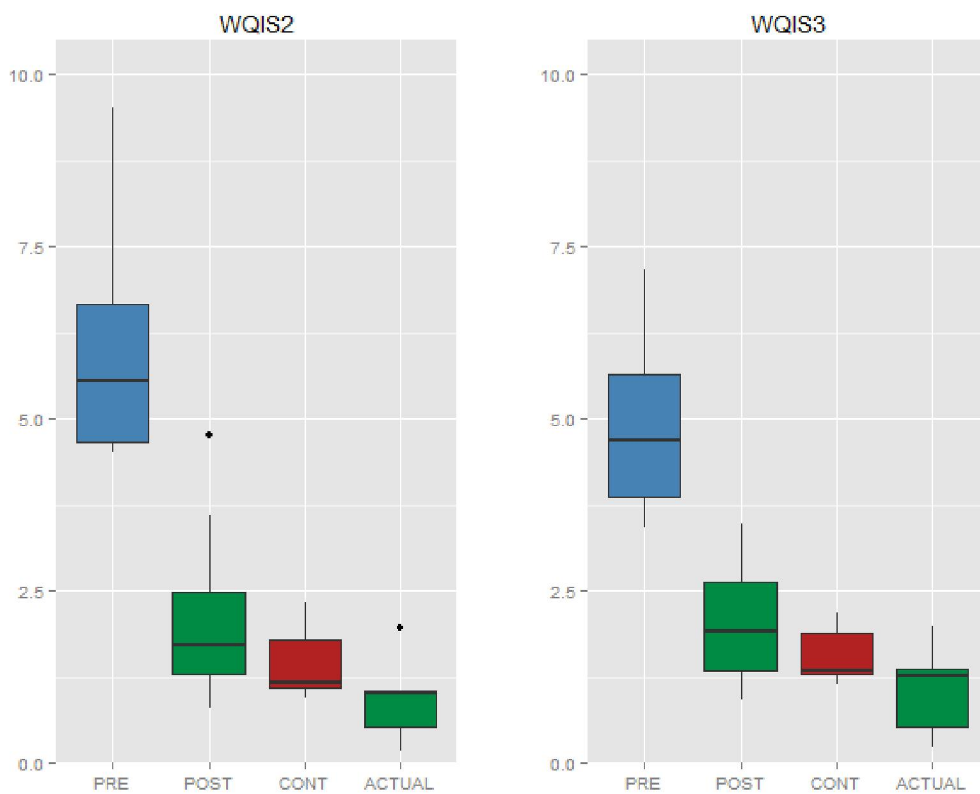


Figure 3-4: Cumulative sum of elevation drops occurring during the Rainbow Trout Flow period for Pre (1984 – 1991), Post (1992-2007), and Continuous (2008-2014) flow years at each water quality index station. Different colours within each graph for Pre, Post and Cont datasets indicate statistical significance ($p<0.05$) as determined by a permutation ANOVA. The "Actual" dataset is included to illustrate variability between predicted CONT values and actual elevation field data collected during 2008 -2014.

3.2 Physical and Chemical Characteristics

3.2.1 Water Temperature

As with the elevation data, 2014 water temperature data also had data gaps, most notably at WQISI and WQ C2 (**Figure 3-5**). Water temperatures during 2014 at the five LCR WQIS varied seasonally, ranging from approximately 3 to 19°C. Temperatures in Kootenay River (WQ C2) were slightly higher, and ranged from approximately 3 to 20.2°C.

The 2014 summer daily temperatures were very similar to the mean temperatures recorded during previous years of the study. Water Quality Index 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.

As expected, water temperature followed a seasonal pattern. During MWF flows (Jan 1 – Mar 31), the 2014 water temperatures had very little variation and were typically between 4 and 5 °C. Temperatures during the RBT flow period (Apr 1 – Jun 30) steadily increased from approximately 4 to 14 °C. Finally, the FFF period exhibited the opposite trend with water temperatures declining from approximately 18 to 10 °C.

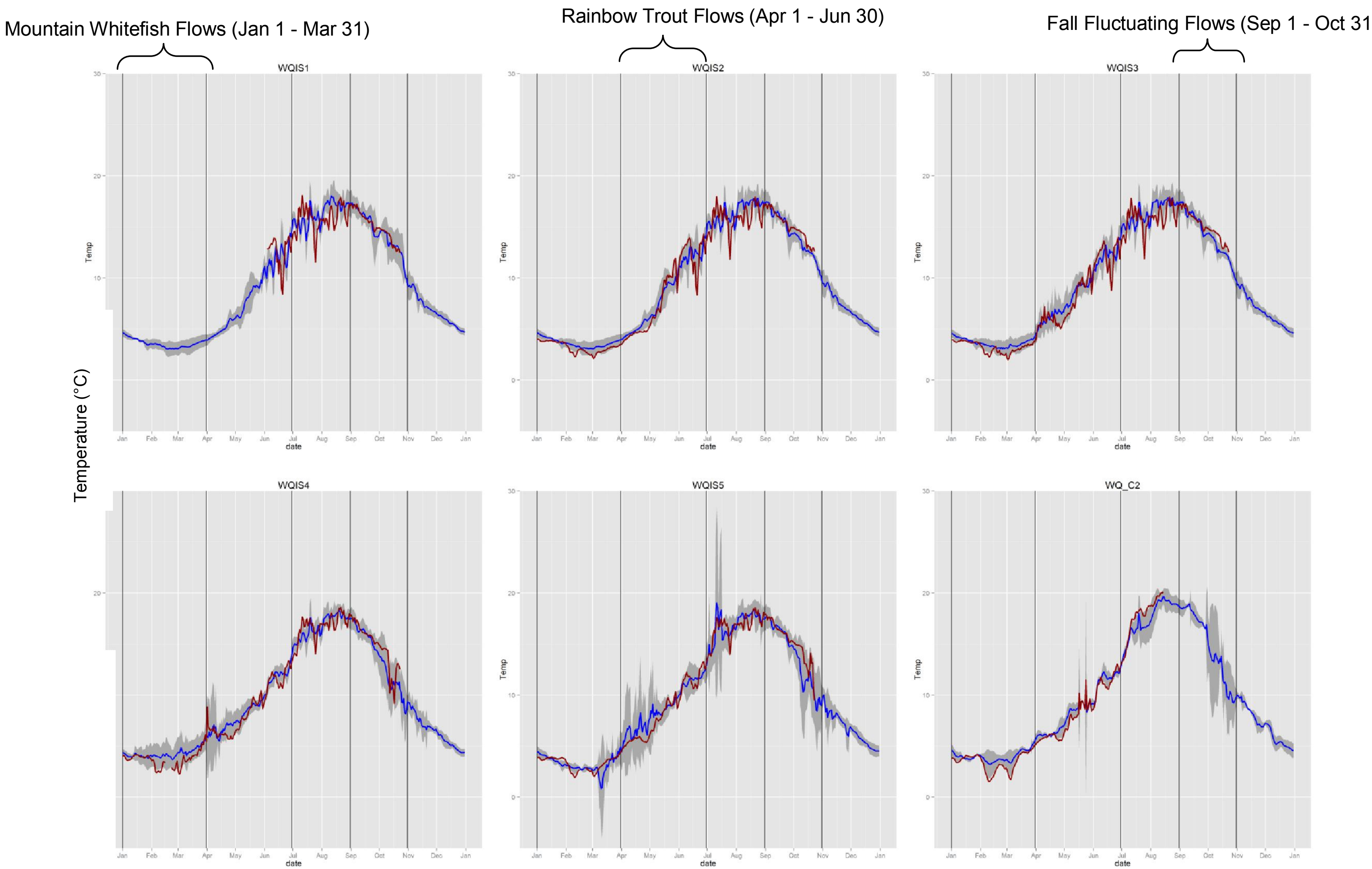


Figure 3-5: Mean daily water temperatures recorded at WQIS1 – 5 on LCR and at WQ C2 on Kootenay River. The red line depicts the mean daily water temperature recorded at each site in 2014. The blue line is the mean daily water temperature throughout the duration of the study (2008-14) ± SD (gray shaded 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 temperature, linear mixed-effects modeling described in Section 2.5.2 of the methods was used. We hypothesized that water temperature may be dependent on the temperature of source waters, air temperature, and the influence of water elevation in upstream reservoirs, and therefore these datasets were also included in the models. This approach allowed us to rank the relative importance of flow regime with other pertinent parameters that may affect water temperature.

This is the first year that separate models were generated for each flow period. The resulting models contained all combinations of explanatory variables. There was only one plausible model for the MWF and RBT flow periods and four plausible models (Δ AICc < 3) for the FFF period. The models explained a high proportion of the variance in LCR water temperature (R^2 = 0.58 – 0.61). Not surprisingly, LCR water temperatures were most strongly correlated with Castlegar air temperature and reservoir water temperatures when all flow periods were considered (**Figure 3-6** and **Appendix A-2**).

Reservoir elevation had a negative effect on LCR water temperature, particularly during the winter and fall, when LCR temperatures decreased with increased reservoir elevation. The effect of flow on river temperature was less, but positively associated during the FFF period and negatively associated during the MWF and RBT flow periods. Based on this analysis, flow is not the most important determinant of river temperature. Reservoir temperature and air temperature were much stronger predictors of LCR water temperature (**Figure 3-6** and **Appendix A-2**).

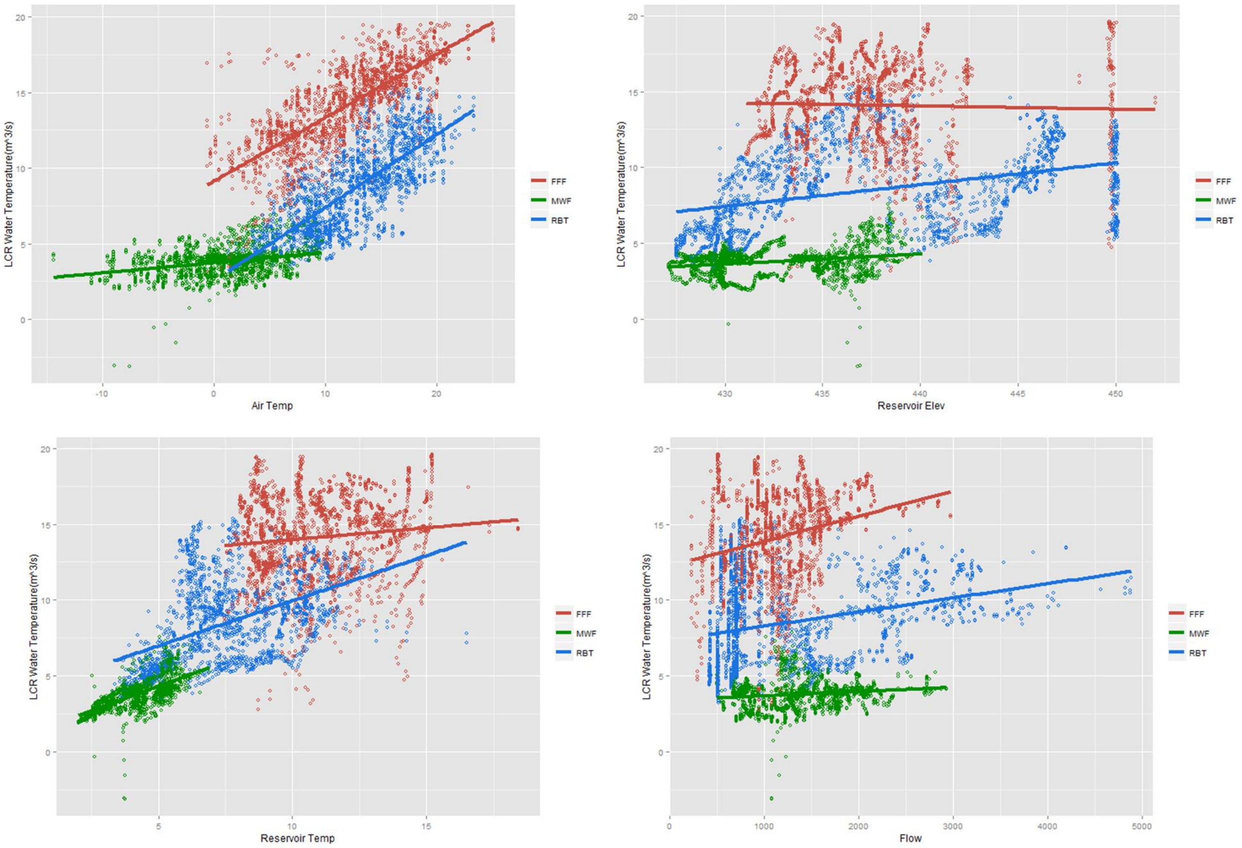


Figure 3-6: Single Linear Regressions of Air Temperature, Reservoir Elevation, Reservoir Temperature and Flow on LCR Water Temperature in Each Flow Period. Flow periods are defined as FFF = Fall fluctuating flows (red), MWF = Mountain Whitefish flows (green), RBT = Rainbow Trout flows (blue).

3.2.2 Water Quality

Water quality sampling for this project has been on-going from 2008 to present. Since 2012, sampling was modified from the previously collected monthly samples in the June to October growing season to allow sampling to be more disbursed over the year and to achieve an overlap with the MWF flow period. Samples were collected on March 28, June 4, August 14, and October 23, 2014. The 2014 results were combined with the entire water quality data set to date. These results are displayed as boxplots according to season: winter (Jan 1 – Apr 5), spring (Apr 6 – Jun 30), summer (Jul 1 – Sep 30), or fall (Oct 1 – Dec 31) (**Figures 3-7** to **3-14**).

The spring 2014 samples were collected on the rising leg of freshet, while the peak freshet occurred four weeks later.

3.2.2.1 Summary of 2014 Water Quality Parameters

3.2.2.1.1 pH

Over the years, pH values have occasionally exceeded the LCR upper pH objective limit of 8.5, but remained below the BC MOE guideline for aquatic life of 9.0. During all four seasons of 2014, mean LCR pH was low throughout LCR compared to typical values, and was particularly low in summer at WQIS1 – WQIS3, and in fall at WQIS3 (**Figure 3-7**). Since this is field meter pH, calibration can drift. pH at mainstem LCR sites averaged 7.8 ± 0.53 (SD) and ranged from 7.3 – 8.9, with the highest values recorded in summer 2014. For reference, pH in the Arrow Lakes Reservoir upstream of the HLK dam ranged from 7.87 – 7.98 in Apr-Nov 2014 (BC MoE data).

In 2014 pH results, only one value exceeded 8.5 (Summer WQSI1; **Figure 3-7**). pH at mainstem sites above the confluence with Kootenay River ranged from 7.5 - 7.6 ± 0.27 in 2014. pH in the mainstem of LCR increased below the confluence with Kootenay River in every season. Winter pH at the sites above the Kootenay confluence averaged 7.8 and were similar to pH measured in other seasons. The lower pH objective of 6.5 has not been exceeded in the LCR during this study.

Photosynthesis raises pH and increased summer pH in the Kootenay River from a minimum of 7.8 in spring to a maximum of 8.2 in summer 2014.

Throughout the study period, Norns Creek exhibited the widest range of pH, likely due to source flows originating from a smaller watershed that has low carbonate buffering capability. Kootenay River showed the narrowest pH range of all the sample sites.

In summary, both the Kootenay and Columbia systems show alkaline and stable pH. All 2014 LCR pH values were within the BC MoE Guidelines.

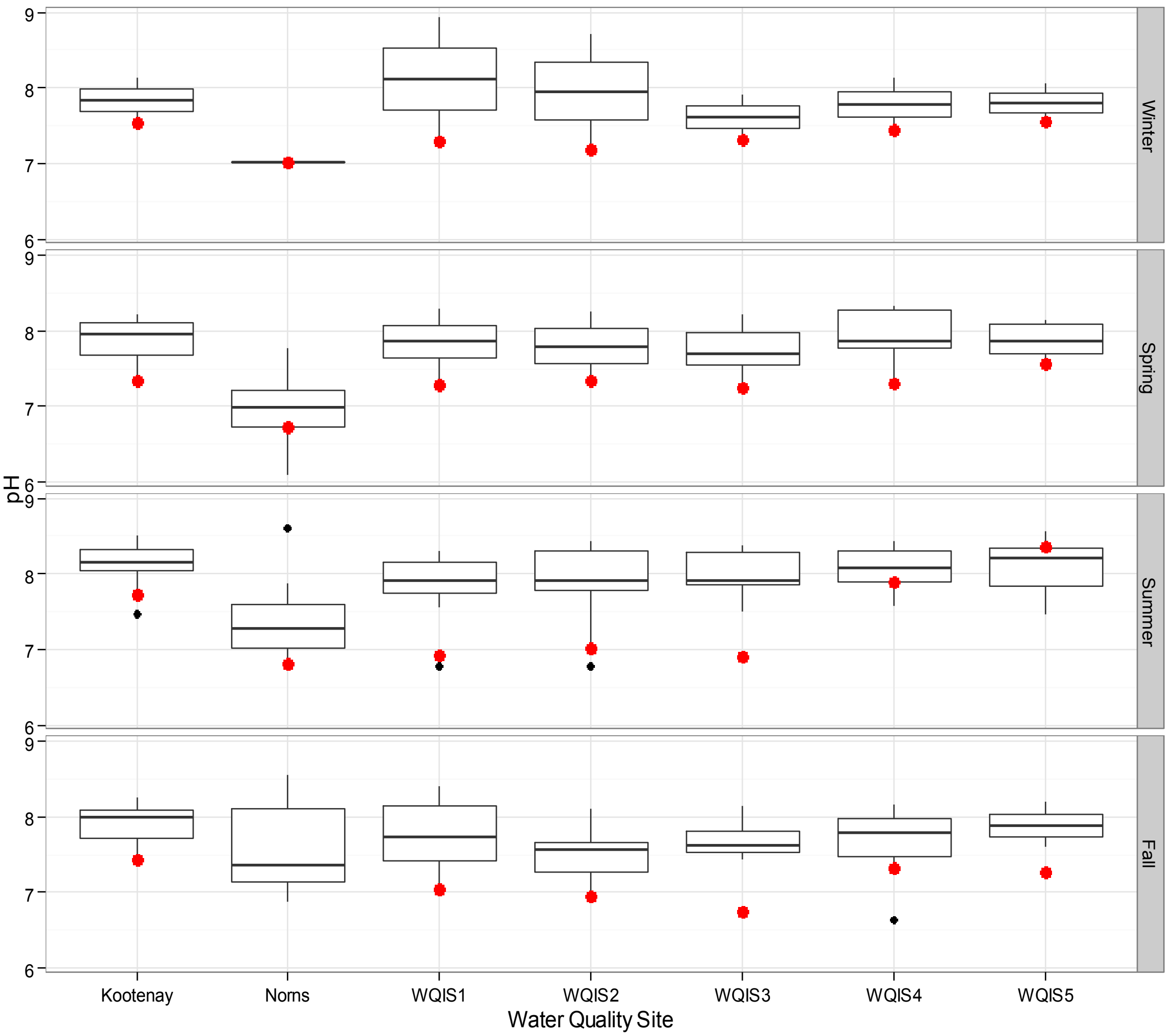


Figure 3-7: pH from LCR Water Quality Index Sites and Main Tributaries (2008-2014). 2014 data is shown in red. The LCR lower and upper pH objective limits are 6.5 and 8.5, respectively, and the maximum BC MOE guideline for the protection of aquatic life is 9.0.

3.2.2.2.2 Electrochemistry Parameters

Specific conductance, total dissolved solids (TDS), alkalinity and hardness all measure the concentrations of ionized constituents in water and they frequently trend together (**Table 3-3**). There is some overlap in the measured ions. For example, hardness and conductivity both include calcium. Conductivity and TDS were measured by field meter at every site on all trips. TDS was also analyzed at Caro Labs, while selected samples were submitted for alkalinity and hardness analyses.

Table 3-3: Ions Contributing to Electrochemistry Parameters

Parameter	Equation or Principle Ions Measured
Alkalinity	Alkalinity = $[HCO_3^-]_T + 2[CO_3^{-2}]_T + [B(OH)_4^-]_T + [OH^-]_T + 2[PO_4^{-3}]_T + [HPO_4^{-2}]_T + [SiO(OH)_3^-]_T - [H^+]_{sws} - [HSO_4^-]$
Hardness	Mainly contributed by Ca Mg, and also Sr Fe Ba Mn
TDS	Soluble salts that yield ions such as: Na+2 Ca+2 Mg+2 HCO3- SO4-2 Cl- NO3- PO4-
Conductivity	Mainly contributed by CaCO3; also (H+ Ca+2 Mg+2 K+ !\la+2 Cl- S04-2 N03- HCO-, OH-

Electrochemistry parameters found in LCR are comparatively low and are far below the values where direct harm to fish can occur (Butcher 1992, CCME 2012).

Specific conductance was monitored using a field meter. Historically in both LCR and its tributaries, specific conductance showed an inverse relationship with flow. On average, the spring freshet of moderate flow years 2013 and 2014 had higher conductivity readings than in the high freshet years 2011 and 2012. This was probably the result of dilution of base flows during the record freshet years. Conversely, in years with lower dam releases, reduced dilution of base flows including groundwater would result in higher conductivity. Conductivity at LCR mainstem sites in 2014 ranged from 75 - 132 µS/cm, with the lowest values usually occurring in the fall (**Figure 3-8**). The 2014 values from the downstream WQIS5 site were within the range of specific conductance measured at Birchbank between 1983 and 1996 (105 – 160 µS/cm) (Holmes and Pommen 1999).

Throughout the study, Kootenay River had consistently higher specific conductance measurements compared to LCR (**Figure 3-8**). In 2014, it averaged 129 ± 13 µS/cm compared to 110 ± 17 µS/cm in LCR samples, and caused a small increase below their confluence. Norns Creek values ranged from a very low 11 µS/cm to 53 uS/cm in 2014, consistent with historic values. The low conductance observed at Norns Creek is typical of streams whose source is mostly snowmelt.

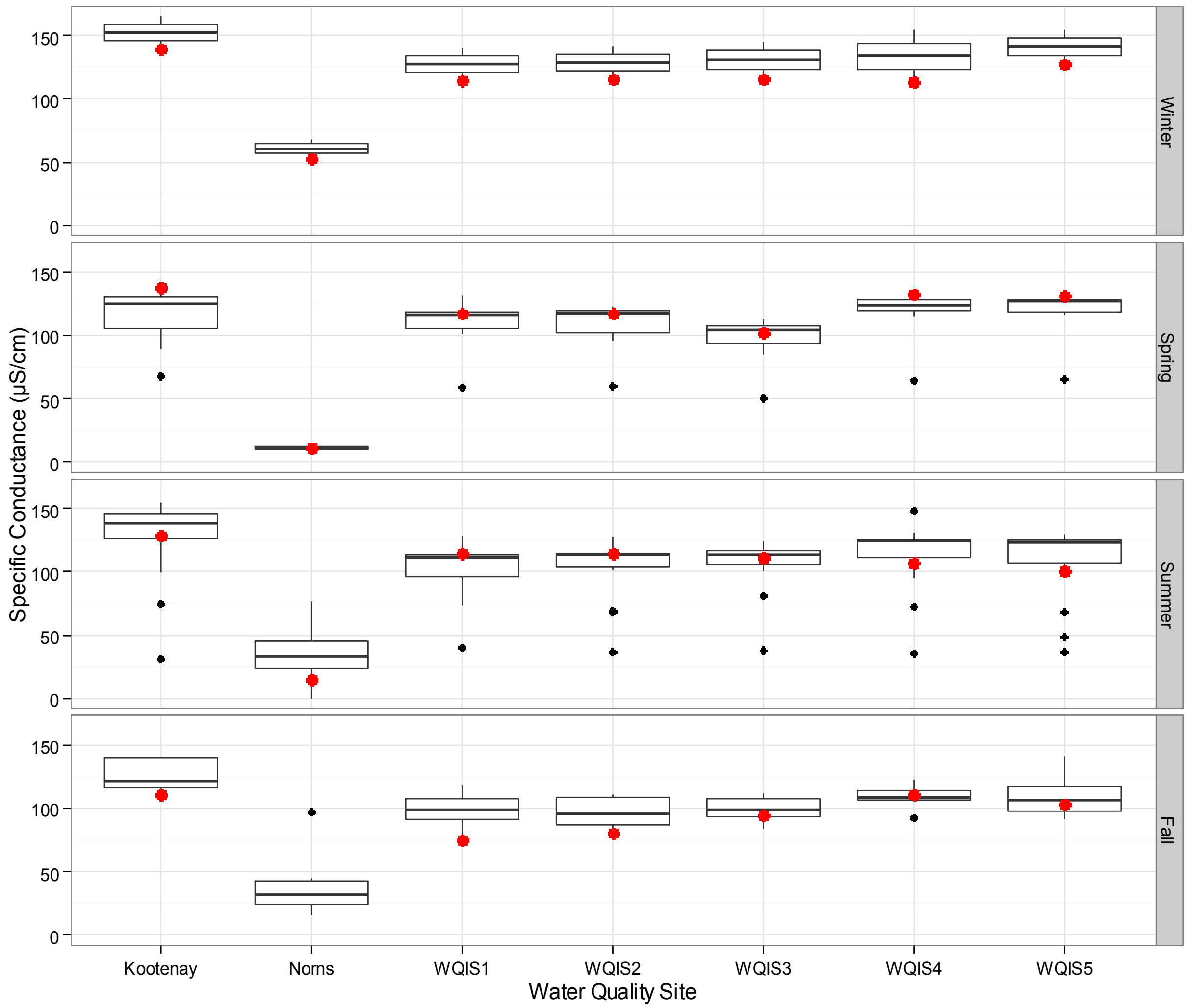


Figure 3-8: Conductivity from LCR Water Quality Index Sites and Main Tributaries (2008-2014). 2014 data is shown in red. No guideline or objective available.

Total dissolved solids results from the lab are shown in **Figure 3-9**. TDS showed similar patterns to field-measured specific conductance throughout the 2014 dataset. TDS averaged 77 ± 9 mg/L and ranged from 55 – 98 mg/L in LCR during 20144. At mainstem sites, TDS was highest in spring 2014, with the exception of WQSI3 where it was highest in winter. Elevated TDS in spring may be the result of spring sampling being a month ahead of peak freshet flows. TDS tended to increase as water travelled through LCR and that increase was most evident in the fall and winter low flow periods.

Consistent with previous years of the study, TDS in Kootenay River usually exceeded that of LCR, and averaged 89 ± 19 mg/L during 2014. The higher TDS observed in Kootenay River was reflected in observed increases in TDS at LCR sites downstream of their confluence in all seasons but winter. This was particularly evident at WQIS4 during the summer and fall seasons for which the most data exists from previous years (**Figure 3-9**).

Norns Creek had consistently lower conductivity and TDS than the mainstem sites, even during very low flow periods such as fall and winter. It averaged 37 ± 12 mg/L during 2014. This low TDS indicates that Norns Creek watershed is dominated by granitic geology (non-carbonate minerals).

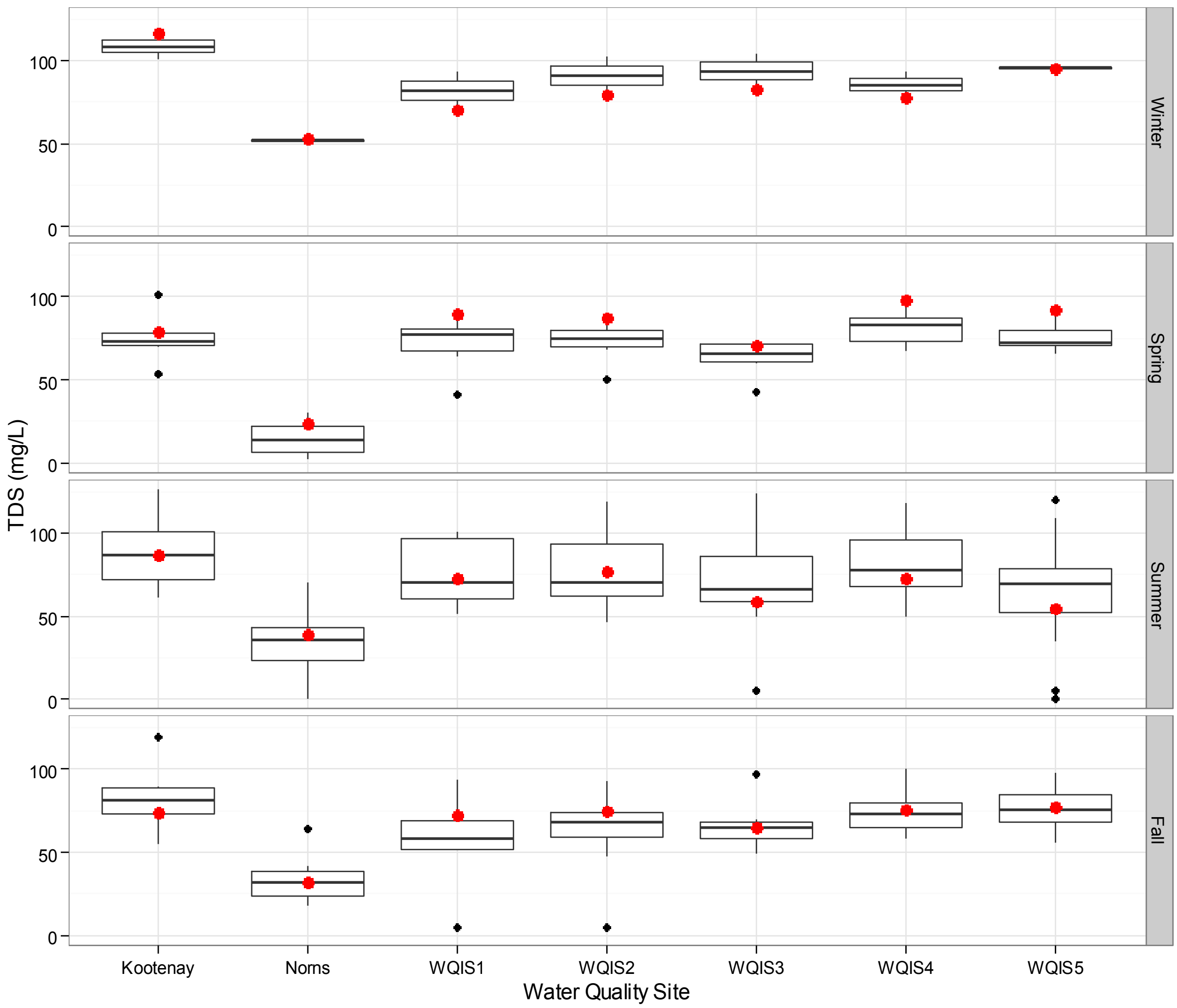


Figure 3-9: Total Dissolved Solids from LCR Water Quality Index Sites and Main Tributaries (2008-2014). 2014 data is shown in red. No guideline or objective available.

3.2.2.2.3 Inorganic Nitrogen

The forms of inorganic nitrogen include nitrate, ammonia and nitrite and these are key macronutrients that are repeatedly consumed, transformed and released as water travels downstream. As is often the case in rivers, inorganic nitrogen is dominated by nitrate throughout the LCR. Nitrate donated by flows from ALR ranged from 0.062 - 0.128 mg/L 0.140 – 0.177 mg/L as N during Apr-Nov 2014 (BC MoE).

All LCR sites, Kootenay River and Norn's Creek were far below the BCMOE aquatic life nitrogen guidelines of 3 mg/L nitrate and 0.7 mg/L ammonia. Similar to previous years, ammonia and nitrite were consistently non-detectable (<0.02 and <0.01 mg/L, respectively) in 2014, as is expected in aerobic environments. There was one notable exception in October 2014, where ammonia measured 0.05 mg/L NH_3 at the sample site WQSI3 adjacent to a City of Castlegar municipal outfall. This sample also had elevated organic N and T-P, suggesting an influence from that outfall.

LCR nitrate concentrations averaged 0.09 ± 0.02 mg/L NO_3 as N overall in 2014, and 0.079 ± 0.02 mg/L NO_3 as N in spring to fall samples, which was 35% higher than the 0.051 mg/L NO_3 as N reported for earlier years of this study. Nitrate concentrations in winter LCR mainstem samples remained above 0.10 mg/L as N inorganic nitrogen (0.119 – 0.127 mg/L), while samples from the balance of the year had concentrations below 0.10 mg/L as N. The percent difference between fall / winter low flows and spring/summer high flows in 2014 was 33% more during low flows for the mainstem LCR sites. This may help explain the high productivity occurring in the LCR during winter.

Nitrate concentrations in the LCR were elevated in the fall at sites closest to the dam, possibly as a result of the fertilization program on the Arrow Lakes Reservoir (Larratt *et al.* 2013) because inorganic nitrogen added as fertilizer should theoretically arrive in LCR after July through October each year (Berube *et al.* 2012), if it has not been consumed within ALR (**Figure 3-10**). The highest concentrations of inorganic nitrogen occurred in the winter low flow period and averaged 0.107 mg/L NO_3 as N in 2014.

During 2014, the Kootenay River nitrate samples averaged 0.099 ± 0.05 mg/L NO_3 as N (**Figure 3-10**). Like ALR, a fertilization program is also active on Kootenay Lake. In winter and spring, Kootenay flows had more nitrate than LCR, while in summer and fall, the reverse was true. The percent difference between fall/winter low flows and spring/summer high flows was 36% more during low flows at the Kootenay site in 2014. In the high flow years 2011 and 2012, Kootenay River had similar nitrate concentrations to LCR during freshet (spring), but declined during the clear flow period (summer and fall). In 2013 and 2014 with moderate freshets, the spring concentrations were elevated to 0.075 and 0.105 mg/L NO_3 as N, respectively.

As with previous years, nitrate concentrations were much lower in Norns Creek than at the mainstem sites, and averaged 0.028 ± 0.04 mg/L NO_3 as N. Amounts of nitrate that would be considered stimulatory to periphyton were only found in the winter low flow period when groundwater inflows would be important to base flows. For the balance of the year, Norns had consistently low nitrates but moderate phosphorus concentrations. Agriculture occurs along Norn's lower length, but did not appear to elevate inorganic nitrogen concentrations.

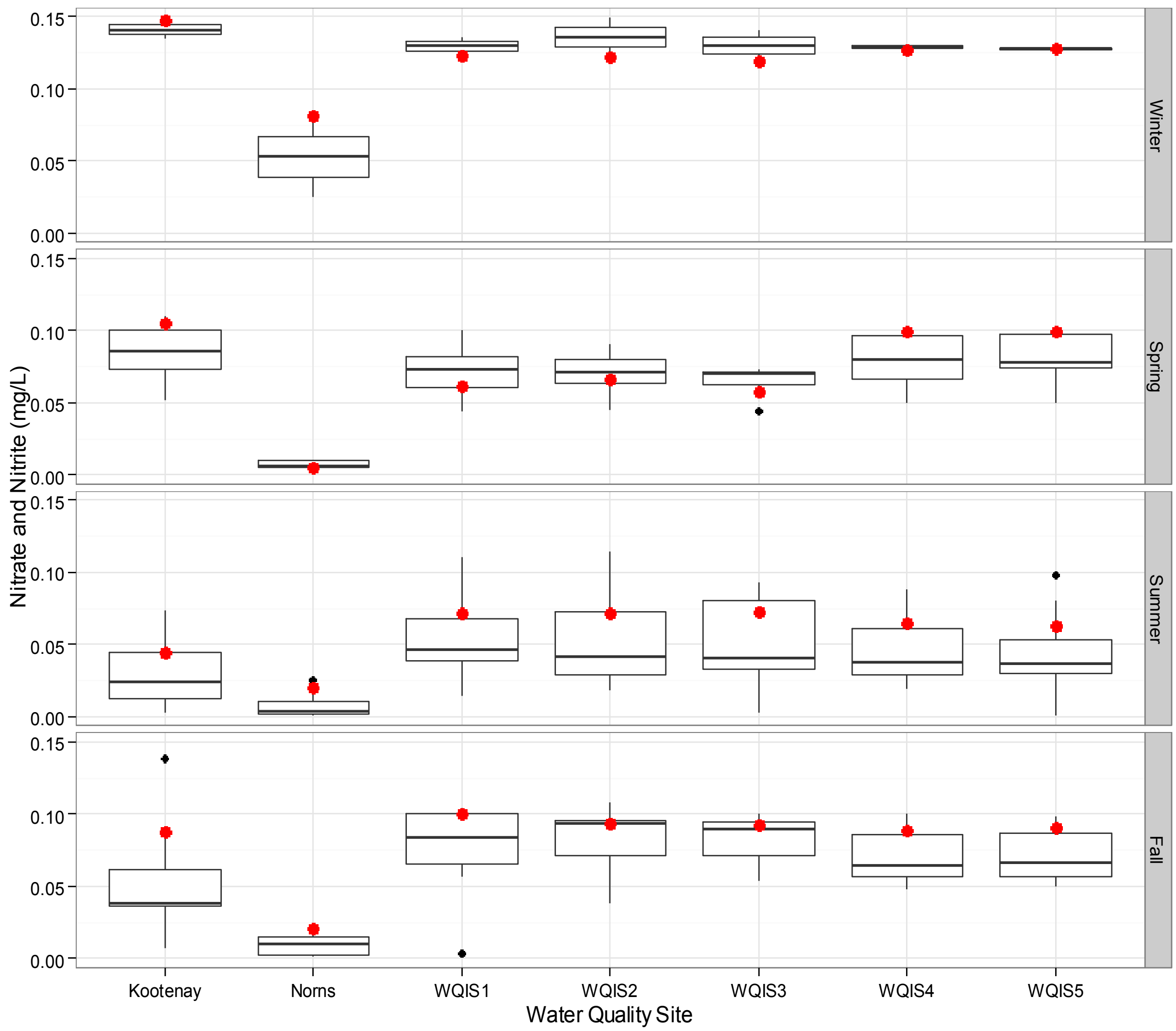


Figure 3-10: Nitrate and Nitrite from LCR Water Quality Index Sites and Main Tributaries (2008-2014). 2014 data is shown in red. BC MOE guideline for the protection of aquatic life is 3 mg/L nitrate; 0.7 mg/L ammonia.

3.2.2.2.4 TKN and Organic Nitrogen

Total Kjeldahl nitrogen measures ammonia and organic forms of nitrogen which include N in algae, detritus, woody debris, etc. Since ammonia was consistently non-detectable in all LCR data, TKN can be assumed to represent organic nitrogen. Samples were only collected in 2009 and were resumed in 2014, thus a box plot could not be made.

The LCR mainstem sites averaged 0.14 ± 0.04 mg/L TKN as N in 2014 samples. For the mainstem LCR sites, the percent difference between fall/winter low flows and spring/summer high flows in 2014 was 29% more TKN during high flows because they carry more detritus. The Kootenay site averaged 0.18 ± 0.12 mg/L TKN as N in 2014 samples. The percent difference between fall/winter low flows and spring/summer high flows was 46% more TKN during high flows in the Kootenay River during 2014. Norn Creek averaged 0.13 ± 0.10 mg/L TKN as N in 2014 samples, and it was the lowest of the sampled sites again in 2014.

Total N can be calculated as nitrate + nitrite + TKN, with TKN (organic N) being the largest component in LCR. The mainstem sites averaged 0.178 ± 0.02 to 0.271 ± 0.116 mg/L T-N as N with the lowest total N at WQSI1 and the highest at WQSI3. The highest readings at each site were variable by season. Total nitrogen donated in flows from ALR to LCR ranged from 0.147 to 0.216 mg/L in the upper 20 m (spill and release at HLK) during Apr-Nov 2014 (BC MoE data). This range is lower compared to LCR mainstem sites in 2014. Taken together, these results suggest that there are additional nitrogen sources in the LCR that augment concentrations in the flows from ARL above its confluence with Kootenay River. Kootenay River had high T-N concentrations at 0.274 ± 0.083 mg/L T-N while Norns Creek was low at 0.153 ± 0.110 mg/L T-N as N.

3.2.2.2.5 Phosphorus

Total phosphorus (T-P) represents the sum of dissolved and particulate phosphorus in a water sample. In addition to biologically available SRP, total phosphorus can include P tied up in algae, organic phosphates, P-bearing minerals and P adsorbed onto mixed phases (e.g. clays, organic complexes, metal oxides and hydroxides) (Maher and Woo 1998).

The recommended maximum SRP to avoid excessive algae growth in rivers is 0.05 mg/L as P (Bowes *et al.* 2010) while the maximum recommended total phosphorus concentration is 0.03 mg/L as P (PWQO 2005). All LCR ortho-phosphate and T-P concentrations were well below these thresholds and indicate oligotrophy. However, biologically important quantities of SRP are probably still present in the LCR as indicated by its stable, diverse periphyton populations.

The range of total phosphorus in ALR was <0.002 – 0.0034 mg/L in the upper 20 m during Apr – Nov 2014 (BC MoE data). This range is wider and higher than the one determined for LCR of 0.002 – 0.018 mg/L T-P (Figure 3-11). However, the T-P means in 2014 were 0.003 ± 0.001 mg/L for ALR and 0.006 ± 0.004 mg/L for the mainstem LCR sites above the Kootenay confluence, suggesting greater nutrient concentrations in the LCR.

Total phosphorus concentrations in LCR, Kootenay and Norns appeared to be slightly elevated during summer 2014 when compared to previous years (**Figure 3-11**); however, T-P values were within the range of data observed since 2008. The summer and fall concentrations during previous years of the study (2008-2012) averaged 0.005 ± 0.002 mg/L, compared to 0.007 ± 0.007 mg/L T-P as P in LCR during 2014. An abnormally high outlier at 0.018 mg/L T-P as P occurred at WQIS3 near a municipal outfall in fall 2014. This sample could have been biased by inclusion of organic material in the sample water. Similarly, another outlier occurred in the Norn's Creek fall sample at 0.030 mg/L T-P as P. Recent rains made Norns flows turbid, and it was very difficult to get a clean creek sample without detritus in it. This sample measured at the 0.03 mg/L T-P guideline. Overall, Norns Creek averaged 0.014 ± 0.011 mg/L T-P as P during 2014, and was the highest of the sites sampled in this study. This nutrient concentration may relate to agriculture and may have relevance to its fisheries.

2014 was the second year that data was collected in the winter. The concentrations of total phosphorus in the winter in LCR were variable between WQIS1 and WQIS3 (Figure 3-11). Operations such as Celgar and/or sewage outflows near these locations may affect the range in values observed during winter 2014.

The Kootenay River site averaged 0.008 ± 0.002 mg/L T-P as P during 2014. Total Phosphorus concentrations downstream of the Kootenay confluence reflected its concentration in a given season.

Inorganic ortho-phosphate (or SRP) represents the fraction of T-P that is readily available to periphyton for growth. In 2011 to 2013, SRP never exceeded the detection limit of 0.01 mg/L, except at WQIS4, which is downstream of the Kootenay confluence and several municipal outfalls. Similarly, SRP never exceeded the detection limit at any sample site during 2014, including Norns Creek and Kootenay River. Similarly in ALR, ortho-P seldom exceeded detection, except in July 2014 when it measured 0.013 – 0.019 mg/L as P.

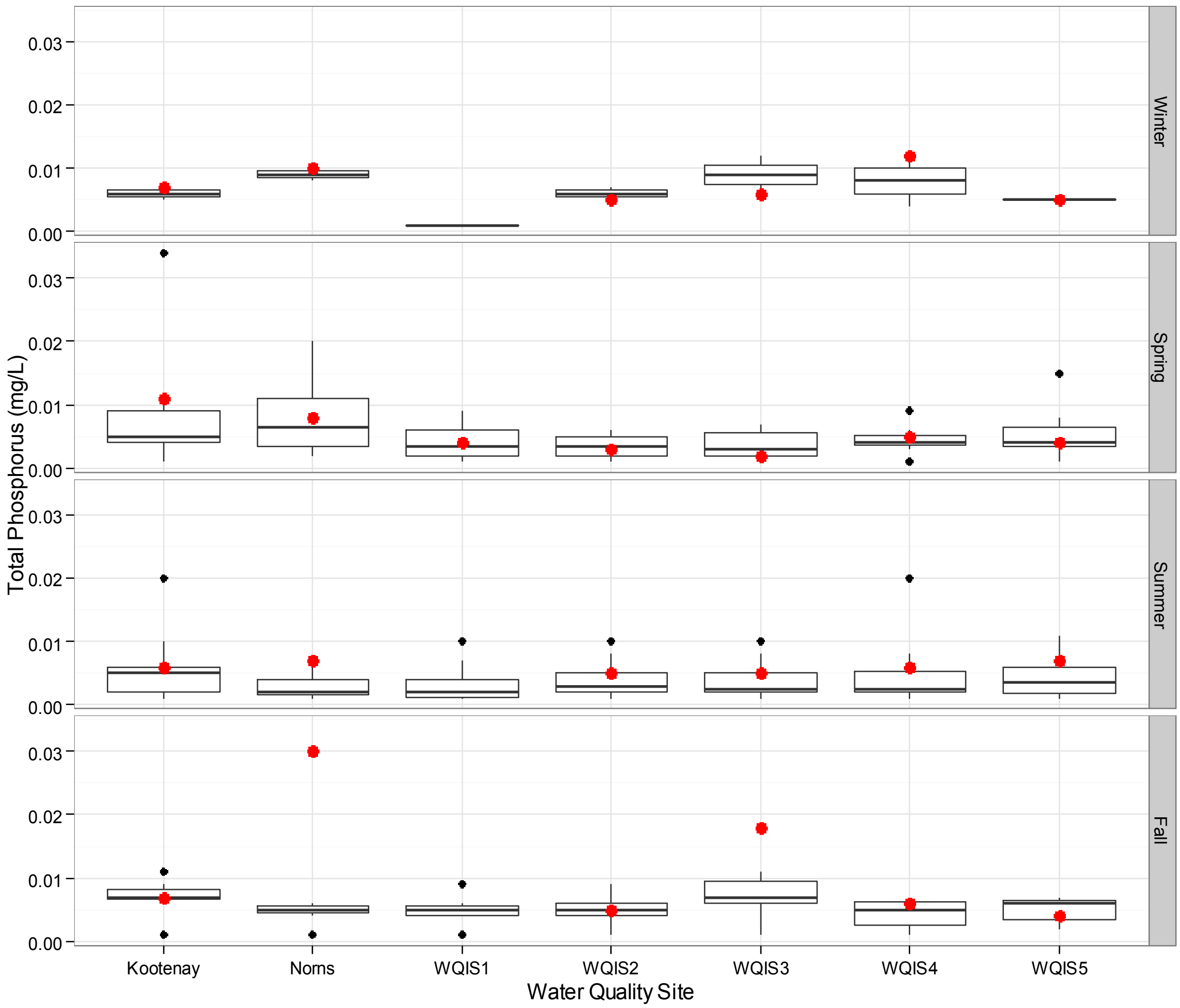


Figure 3-11: Total Phosphorus from LCR Water Quality Index Sites and Main Tributaries (2008-2014) 2014 data is shown in red. BC MOE guideline is 0.005 - 0.015 mg/L for lakes; tentative river guideline = 0.03 mg/L T-P to avoid excessive algae growth.

3.2.2.2.6 Turbidity

Turbidity measures how much sediment, organic detritus and organisms suspended in the water decreases its clarity. The range of turbidity measured in Kootenay and LCR flows was narrow. In LCR, turbidity collected in 2014 was within the range of previous years of 0.3 to 0.9 NTU (**Figure 3-12**). The average LCR turbidity in 2014 was 0.0.46 ± 0.10 NTU. No turbidity spikes of the >7 NTU magnitude seen in past years were observed in 2014. However, it is possible that a freshet turbidity spike was missed since 2014 spring data was collected on June 4th, and the freshet peak flows occurred four weeks later.

Turbidity measured in this study met BC guidelines protective of aquatic life. A turbidity spike would have to exceed background by 2 NTU for 30 days during clear flows or exceed background by 5 NTU at any time when background is 8 – 50 NTU during high flows to exceed the guideline (BC MoE 2012). In a low turbidity regulated system like LCR, it is unlikely that these guidelines would be exceeded, even in peak freshet flows.

Although turbidity can be expected to decrease with settling behind a dam, the turbidity range measured in ALR during Apr-Nov 2014 was 0.14 NTU in deep water to 0.38 NTU in shallow water (BC MoE data). This range is actually lower than the first station on LCR (0.3 – 0.6 NTU) suggesting that there are turbidity sources within LCR (**Figure 3-12**).

As expected, the turbidity at Norns Creek was consistently higher in the spring compared to Kootenay River and LCR. Because the rivers are fed from reservoirs that allow settling of suspended materials, it logical that the turbidity values would be lower than unregulated Norns Creek flows. Kootenay flows averaged 0.53 ± 0.15 NTU, while Norns Creek averaged 1.7 ± 1.3 NTU in 2014. The highest turbidity in Norns was still moderate at 3.5 NTU in October, immediately following a storm.

Turbidity and TSS affect light penetration, particularly into deep water. At the moderate turbidity levels found in LCR, light penetration to the shallow substrates would not hinder photosynthesis (Caux *et al.* 1997; ENSR 2001). However, light penetration through water deeper than about 4 m would be reduced enough to influence periphyton production.

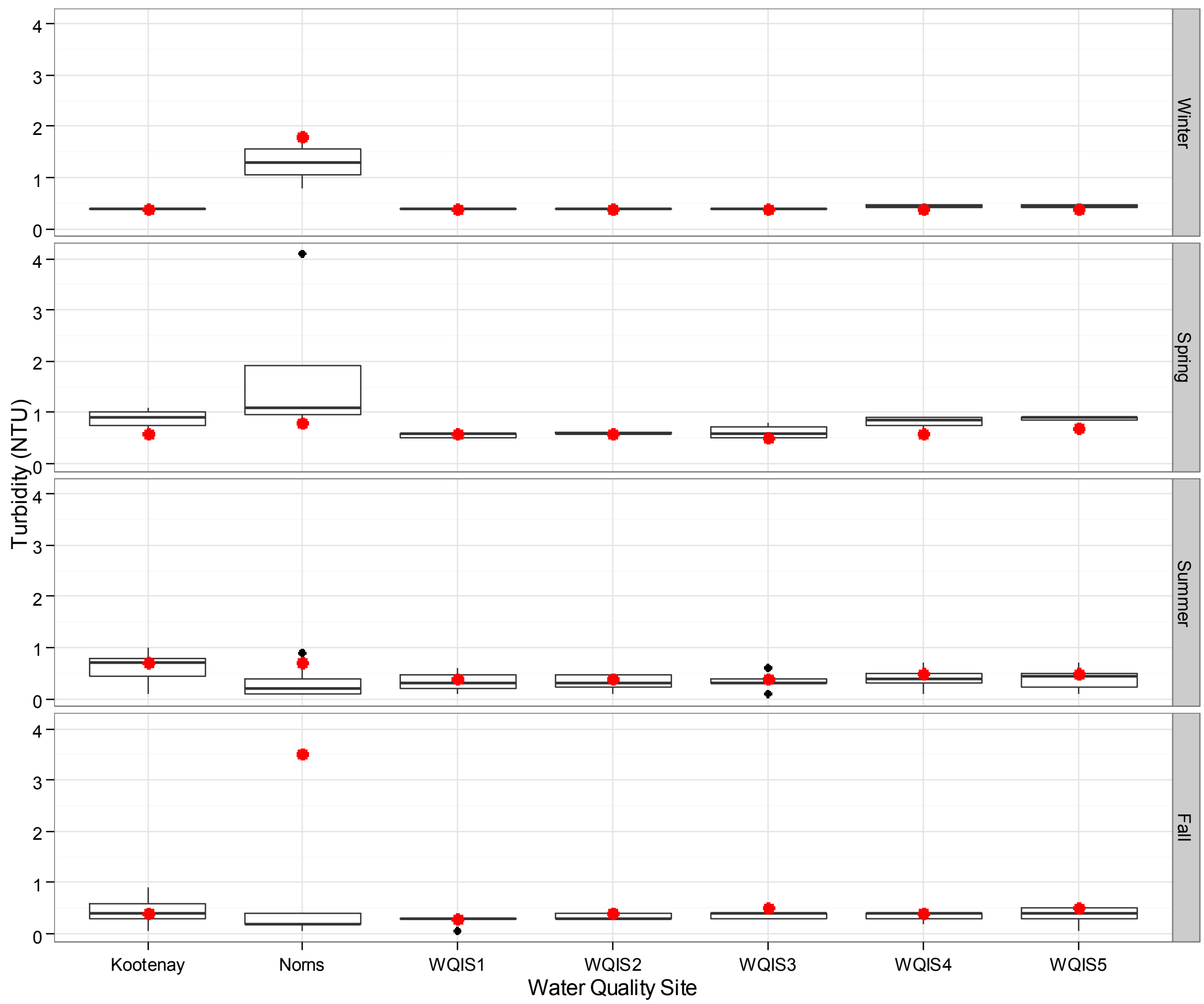


Figure 3-12: Turbidity from LCR Water Quality Index Sites and Main Tributaries (2008-2014). 2014 data is shown in red. Outliers (n=3) were removed to improve plot aesthetics. Aquatic life protection guidelines state maximum 24 hr increase = 8 NTU; maximum clear flow average (30 days) increase = 2 NTU.

3.2.2.2.7 Total Suspended Solids

Total suspended solids (TSS) or non-filterable residue is related to turbidity but this parameter provides an actual weight of the particulate material present in the sample. The relationship between turbidity and TSS depends on the nature of the solids.

Since all mainstem samples consistently had TSS of less than 5 mg/L (**Figure 3-13**), a TSS spike of 25 mg/L for a duration of 24 h in clear flows, or an increase of 5 mg/L for a duration of 30 days in clear flows, could only occur in catastrophic flood. Like turbidity, the BC guidelines protective of aquatic life for TSS are unlikely to be exceeded at LCR mainstem sites.

Total suspended solids concentrations are typically low in the regulated LCR and Kootenay systems and they were very low in 2014. Most samples were non-detectable (< 1 mg/L) with only two exceptions. The Norns Creek sample collected on October 23 2014 measured 11 mg/L TSS because sampling occurred during a large storm. The Kootenay River sample from August 14 2014 measured 2 mg/L TSS which may have been caused by suspended algae.

Higher flows associated with freshet were likely the contributing factor to TSS variability, but it is interesting that higher values were not recorded at all sites in heavy freshet years. To date, overall TSS was higher in Kootenay River, Norns Creek and at WQIS4 and WQSI5 in LCR. The highest recorded values were 3 and 4 mg/L recorded below the Kootenay confluence at WQIS5 and 4, respectively (**Figure 3-13**). Although these values were higher than what was documented in other seasons, they were not necessarily unusual given the higher flow period. As an example, in 2012 the concentration recorded during the spring at Norn's Creek was 15 mg/L. Similarly, recent rains in Norns during the fall 2014 caused a reading of 11.5 mg/L TSS.

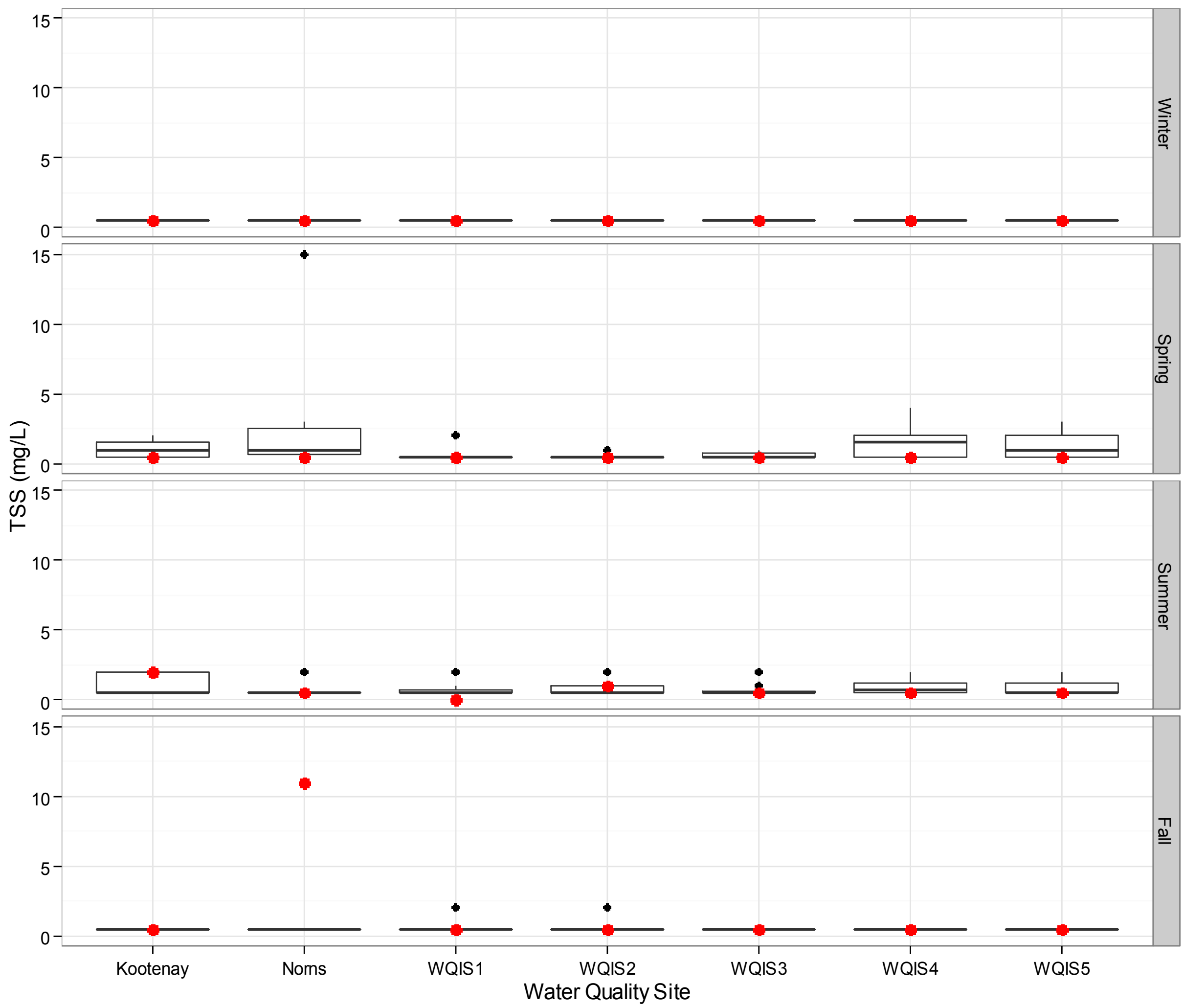


Figure 3-13: Total Suspended Solids from LCR Water Quality Index Sites and Main Tributaries (2008-2014). 2014 data is shown in red. A number of measurements were estimated based on lab tolerance limits. No guideline or objective available.

3.2.2.2.8 Dissolved Oxygen

Dissolved oxygen enters water through turbulent flow, gas exchange and by photosynthesis. The capacity of water to hold dissolved oxygen is a function of its temperature.

Dissolved oxygen concentrations were adequate for all salmonid life stages throughout this study (BC MoE 2012), and usually exceeded the 10 mg/L DO objective set for LCR (Butcher, 1992), even in warm summer low flow conditions.

Dissolved oxygen in LCR mainstem sites ranged from 9.0 – 11.6 mg/L in 2014; a range very similar to previous years. Throughout the study period, dissolved oxygen declined during the summer in response to increased water temperature, but did not fall below the 9.0 mg/L DO Objective at the mainstem sites (**Figure 3-14**).

Dissolved oxygen saturation ranged from a minimum of 89% in fall to a maximum of 113% in spring 2014. Percent saturations above 100% occur naturally with turbulence or when photosynthesis contributes oxygen that super-saturates the water. During this study, dissolved oxygen super-saturation has only been documented in the spring and summer months. The average DO saturation was near 100% during 2014. This was a lower mean than what had been documented in previous years of the study and reflected the shift to sampling during the late fall and winter.

Dissolved oxygen in the Kootenay River during 2014 ranged from 8.6 – 12.0 mg/L with 88% saturation in winter to 116% saturation during spring high flows. These DO ranges are comparable to data collected in previous years (**Figure 3-14**). Norns Creek is the second largest tributary to LCR and in 2014, it measured from 9.4 mg/L in summer to 10.5 mg/L DO in winter and spring, a range within that previously reported (Larratt *et al.* 2013; Scofield *et al.* 2011). Readings were taken from within 1 m of the substrate in Norns Creek and averaged 100% oxygen saturation. All summer 2014 dissolved oxygen samples met the 9.0 mg/L DO guideline.

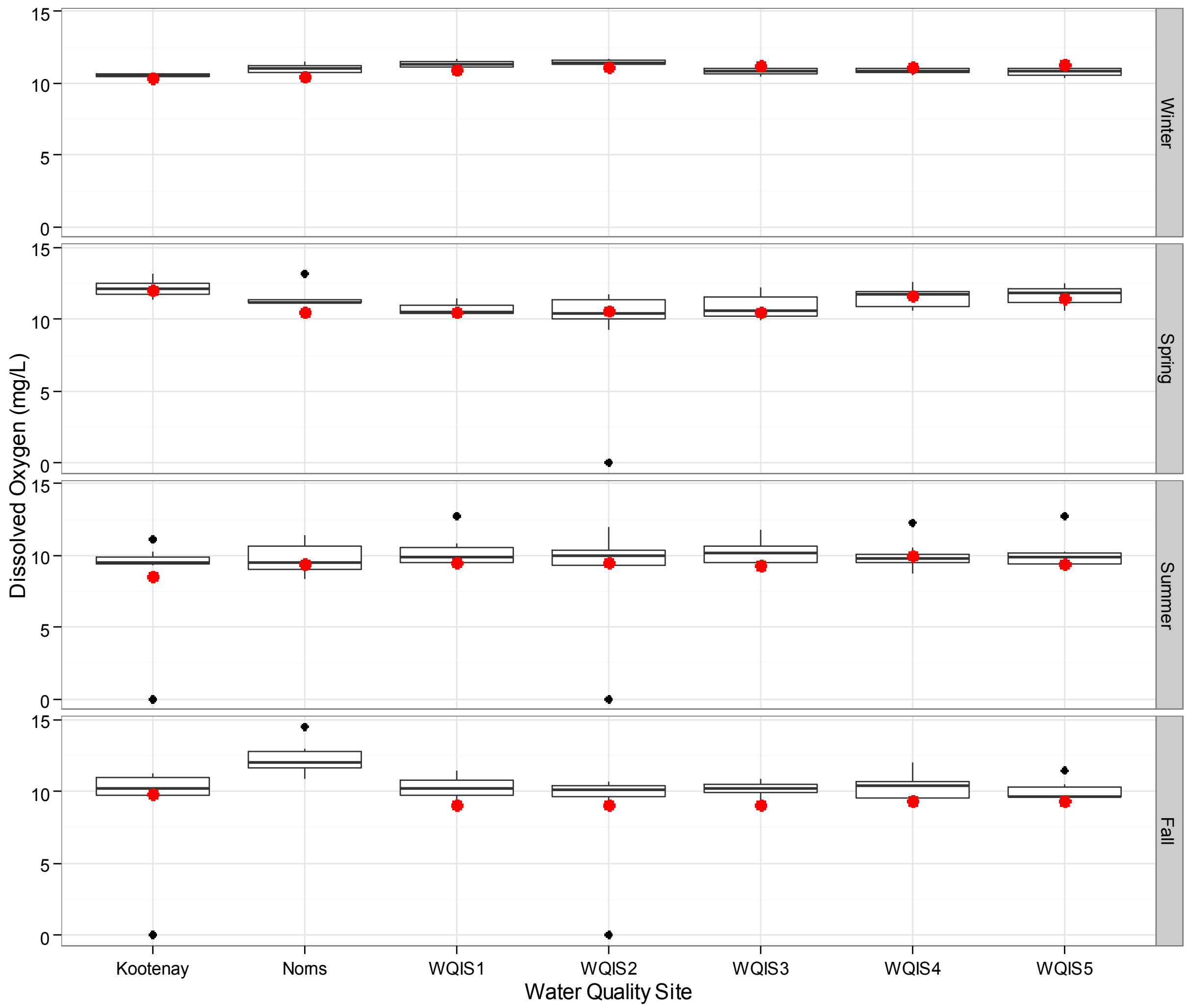


Figure 3-14: Dissolved Oxygen from LCR Water Quality Index Sites and Main Tributaries (2008-2014). 2014 data is shown in red. BC MOE guideline is 9 mg/L; LCR Objective is 10 mg/L.

3.2.2.2 Relationship between Water Quality and Flow

This is our first attempt to statistically test the effect of flow management on LCR water quality. The quantity of available data is minimal and hence the results should be interpreted cautiously. There are many hypothetical interactions that could be important that we have yet to fully explore. Our goal is to use future years of data to better relate the effects of flow regulation, together with other pertinent biophysical factors such as freshet on water quality parameters in LCR.

Testing the hypotheses that implementation of different flow periods (MWF and RBT) does not alter the electrochemistry and availability of biologically active nutrients in LCR involved linear mixed-effects modeling as described in Section 2.5.3. We hypothesized that LCR water quality may be dependent on Arrow Lakes Reservoir (ALR) water quality because previous modelling of LCR temperature showed a direct relationship with upstream conditions (Olson-Russello *et al.* 2014). Data from ongoing nutrient addition in the ALR Nutrient Restoration Program was used with flow management and season variables in the models. A detailed description of each explanatory variable is included in the methods. This approach allowed us to rank the relative importance of flow management with other parameters that can affect LCR electrochemistry and availability of biologically active nutrients.

Model averaging of all WQSI data from all years included the following responses, conductivity, TDS, NO_3 + NO_2, and total phosphorus because these data sets were the most complete. The number of plausible models (those with an AICc<3.0) ranged from 2 to 12 (**Table 3-4**). The models used to generate Appendix A-3 were data-limited due to intermittent LCR water quality sampling. Given this, the initial modeling exercise explored potential relationships between predictor variables and LCR water quality parameters.

Table 3-4: 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 Models

Water Quality Parameter Responses	# of plausible models	range of pseudo R^2
Conductivity	2	0.01-0.52
Total Dissolved Solids (TDS)	12	0.04-0.27
Nitrate +Nitrite (NO_3 + NO_2)	2	0.71-0.74
Total Phosphorus	1	0.70

The nutrient models (NO_3 + NO_2 and total phosphorus) were the most informative, while models of conductivity and TDS described less variation (had lower R^2) and had explanatory variables that were not as useful in explaining the observed trends (RVI's < 0.7) (**Appendix A-3**). Variation in daily flow (Flow Daily SD) was the most important predictor of NO_3 + NO_2 and total phosphorus suggesting that high fluctuations in flow may have a direct effect on biologically active nutrients in LCR. Generally, NO_3 + NO_2 and total phosphorus in LCR increased with greater variability of mean daily flows. This may be caused by factors such as nutrient release from decomposition of organics in the varial zone, or variable groundwater influx.

Operations during MWF and RBT flow periods were also viable predictors for phosphorus, although their importance was considerably less. Note that daily flow variation, elevation difference in the MWF period, and the sum of elevation drops in the RBT period are moderately correlated, so it was not surprising to see these predictors trend together. Interestingly, ALR nutrient addition (total nutrition) and measured nutrients in ALR (dissolved nitrate and total phosphorus) tended to not show a relationship with LCR-measured nutrients. This suggests that the conditions within LCR may be as or more important to biologically available nutrient concentrations.

The conductivity model was less informative than nutrient models, but it identified ALR nutrient addition, spring and summer as important predictors of conductivity. The seasons were negatively correlated with conductivity. This finding is consistent with measured conductivity in LCR. Conductivity tends to have an inverse relationship with flow, therefore in higher freshet years the measured conductivity was lower than in moderate years (Olson-Russello *et al.* 2014).

3.3 Periphyton

Our periphyton sampling effort has been focused on 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

Time series accruals were collected in summer and fall 2008 - 2010 using chlorophyll-a (Scofield *et al.* 2011). This data indicates that accrual time required for LCR periphyton to reach peak chl-a on closed cell Styrofoam was 6-7 weeks in the summer and more than 8 weeks during the fall fluctuating flows. In summer accruals, growth leveled off by 4-5 weeks in all three years of their study. By that time, periphyton losses and gains were matched, perhaps through flow-induced shear and through grazing, as benthic invertebrates were frequently observed on the samplers. In the fall, the growth phase extended beyond 8 weeks, and reached far higher chl-a concentrations than the same period in the summer deployments. Samplers were deployed in winter 2013 for 12 and 26 weeks and chl-a peaked at 12 weeks, although periphyton biovolume continued to climb

(**Table 3-5**). In winter 2014, both chl-a and biovolume were lower at 20 weeks than at 10 weeks on these mid-depth samplers. Accrual data for chl-a was collected in winter 2014 for the first time and it suggested that the accrual time required for LCR periphyton to reach peak biomass during the winter was approximately 10 weeks (**Figure 3-15**). Although data was highly variable between seasons and years, the data to suggest that accrual reaches peak biomass in 6-7 weeks in Summer, greater than 8 weeks in Fall, and in 10 – 12 weeks in Winter. When samplers were deployed for longer than these periods a combination of sloughing, grazing, shading by surface algae layers and bacterial decomposition of algae cells deep in the periphyton biofilm, reduced the standing crop of periphyton in LCR.

Table 3-5: LCR Periphyton Metrics for Mid-Depth Samplers Deployed for 10, 12, 20 and 26 Weeks in Winter 2013 and 2014

Duration/Year	**Chlorophyll-a** $\mu g/cm^2$		**Biovolume** cm^3/m^2	
Sampler Depth	MS	M	MS	M
12 week/2013	10.8	10.9	14.7	23.6
26 week/2013	8.54	5.88	38.5	43.1
10 week/2014	7.6	7.36	9.48	7.9
20 week/2014	2.83	4.79	3.46	3.25

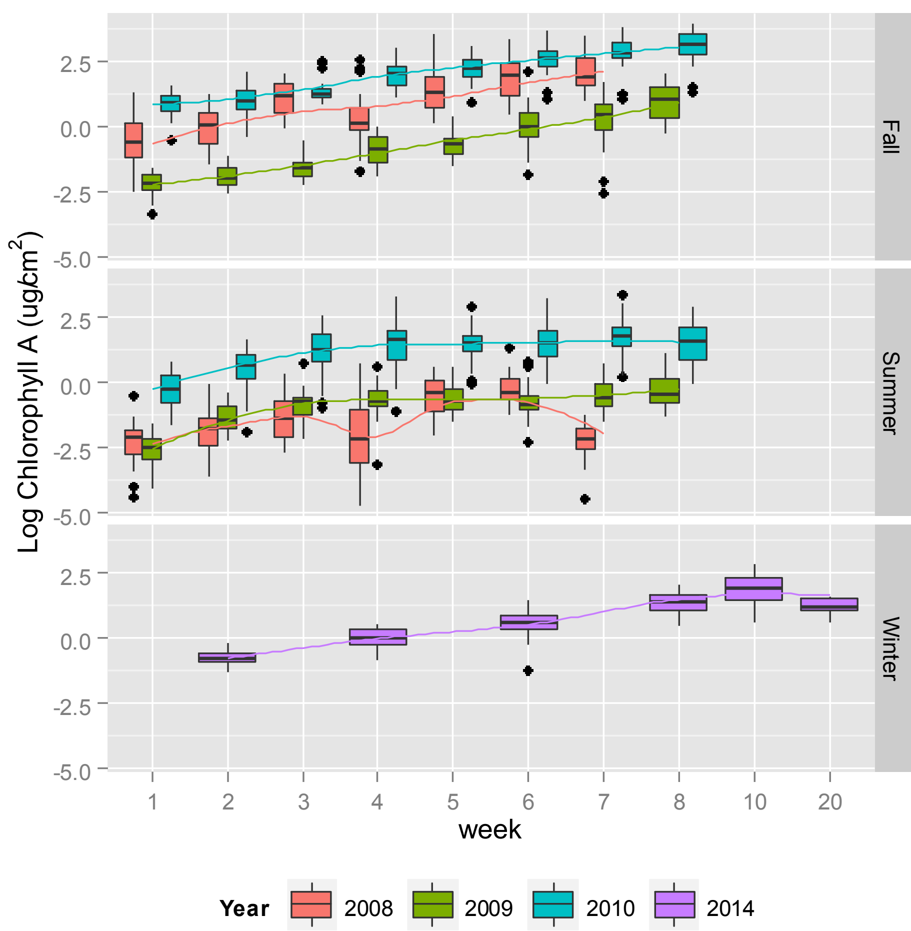


Figure 3-15: Weekly Periphyton Chl-a Accrual Rates (2008 – 2010 and 2014) in the Summer, Fall and Winter. Fitted lines were generated using a locally weighted polynomial regression method (LOWESS). First three years of data were obtained from Scofield *et al.* 2011.

3.3.2 Periphyton Community Analysis

Community analyses of the 2008 – 2014 periphyton data were completed at the genus level to reduce the potential effects of taxonomist and the effects of rare species, allowing focus on large scale trends. NMDS reduced the genus level community compositions to 2 dimensions/axes. The stress index was 0.19, which indicates the two NMDS axes are a good representation of the periphyton community composition. A permutational MANOVA indicated that periphyton community compositions exhibited significant differences when grouped by year (F=67.9, p=0.001), season (F=23.9, p=0.001), depth (F=4.33, p=0.001), and site (F=1.57, p=0.006). However, more variance in periphyton community compositions was explained by year (R^2=0.17) and season (R^2=0.13), whereas transect (R^2=0.05) and site (R^2=0.03) accounted for a smaller amount of variance. The stronger separation of community compositions in terms of season and year is also seen in **Figure 3-16**.

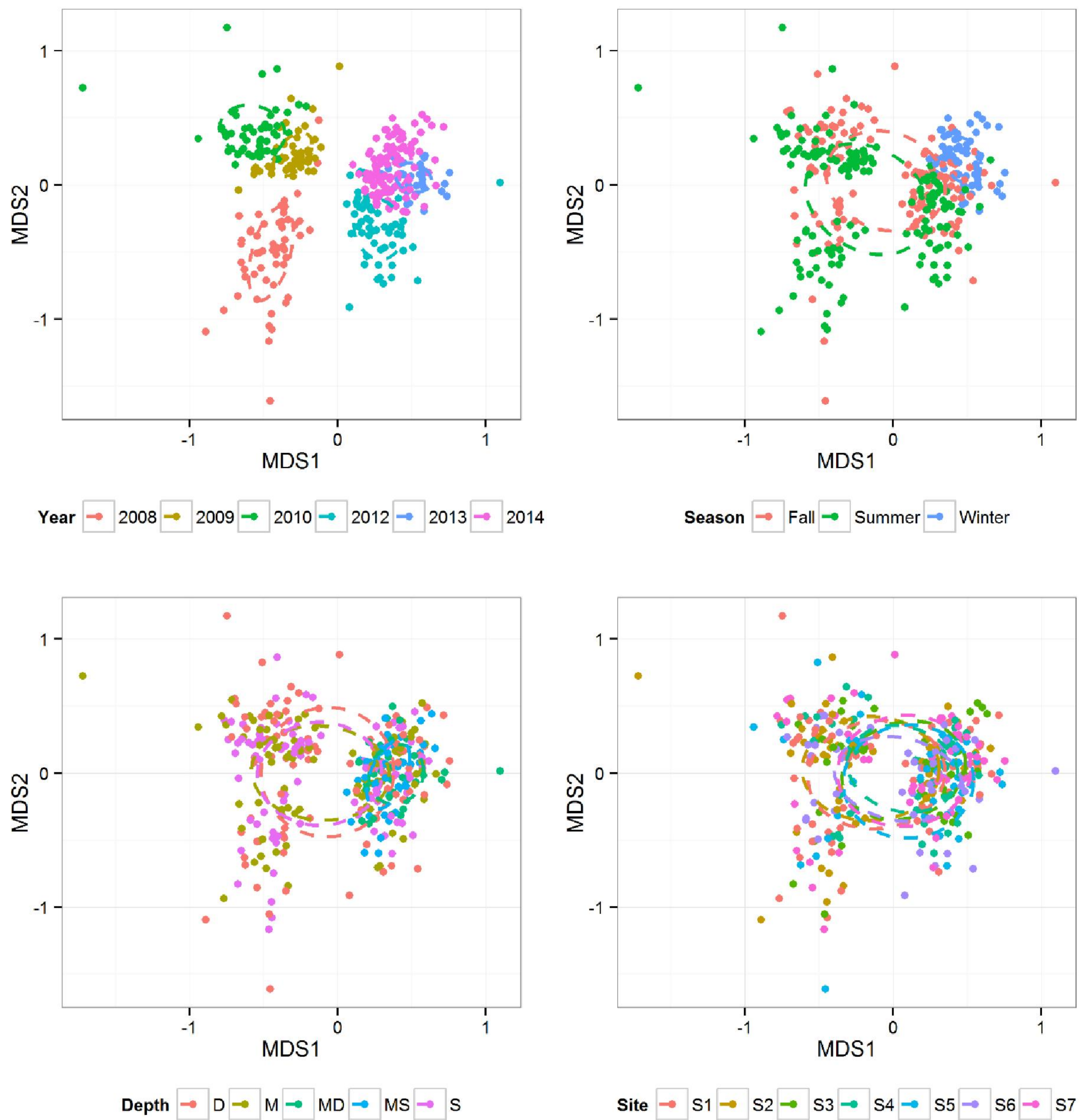


Figure 3-16: NMDS of Periphyton Genus Level Abundance grouped by Year, Season, Depth, and Site for all Data between 2008 – 2014. The NMDS used a Bray-Curtis dissimilarity index and had a stress index of 0.19. Ellipses are calculated based on 95% CI of the NMDS scores for each group.

A total of 60 periphyton algae taxa were frequently observed in LCR. Like most large rivers, LCR periphyton was dominated by diatoms representing between 50 and 100% of the average biovolume in all sample sites to date (**Table 3-6**). Over the years of study, the largest shifts in community structure occurred in the soft-bodied algae. For example, flagellate abundance ranged from 0 – 30% while filamentous cyanobacteria ranged from 6 - 80% by abundance, but that translated to only 0.01 – 8% 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 – 94% of biovolume. Their prevalence was much lower in the winter at 0 – 28%. The nuisance diatom *Didymosphenia geminata* (Didymo) was detected at all LCR sample sites and was most prevalent in winter.

Table 3-6: Range of Periphyton Relative Abundance and Biovolume Obtained from Artificial Substrates by Season and Year

LCR	Summer 2012 - 2014		Fall 2012 - 2014		Winter 2013 - 2014	
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	16 - 83	39 - 99	18 - 88	5 - 100	28 - 93	71 - 100
Flagellates	0.6 - 15	0.1 - 10	0 - 30	0 - 67	1 - 19	0 - 3
Cyanobacteria	6 - 80	0.1 - 3	6 - 80	0.01 - 8	5 - 91	0.02 - 3
Green	0.1 - 72	1 - 60	0 - 33	0 - 94	0 - 6	0 - 28

Large variations in the periphytic species assemblage and production metrics were observed among the years of study. 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.

When all periphyton growth metrics are considered for the 2008 – 2014 sample period, the lowest measured growth occurred in summer 2008 and the highest occurred in fall 2010. Overall, the lowest periphyton production occurred in 2012, the record flood year. All this indicated a strong periphytic response to flows in LCR.

3.3.3 Periphyton Production Models

Model averaging of key periphyton responses included total abundance, biovolume, chl-a, species richness, Simpson's index, percent community from reservoir and percent good forage. 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 2008. For each response, the following explanatory variables were used: added nutrients, flow daily SD (fall only), elev. diff. (MWF) (winter only), elev. diff. (RBT) (summer only), velocity, and substrate score. A detailed description of each explanatory variable is included in the methods (Section 2.5.5). The number of plausible models (those with an AICc<3.0) ranged from 2 to 14 across all seasons and the total number of models considered was 96 for each season (**Appendix A-4**).

Elevation Diff (MWF), a measure of flow variability, was the most important predictor of total abundance in the winter, suggesting that operations may have a direct effect on productivity (**Figure 3-17**). The negative coefficient means that the total abundance of periphyton decreased with an increasing water elevation difference between MWF spawning (Jan 1-21) and MWF incubation (Jan 21-Mar 31). These are the first results reported to date that suggests that the management of the MWF flow period may have an effect on the benthic community. The data is preliminary and additional sampling during the winter is needed to confirm the relationship. Elevation Diff (MWF) was not an important predictor of either biovolume or chl-a during the winter. Flow variability may be affecting species in a specific manner that is not fully understood (i.e., operations may be stripping away more abundant but smaller cells for instance). For biovolume, the most important variable during the winter was substrate score, suggesting that as substrate size increased so did periphyton biovolume. This result is expected since larger substrates are more stable and tend to result in algal communities that have a larger biovolume per cell than communities occurring in areas of finer substrates. All other models tended to explain less variation (i.e., had lower R^2) or had explanatory variables that were not useful in explaining the observed trends (i.e., RVI's < 0.6 or 0.7).

During the summer (RBT flow period), velocity was the most important predictor for abundance, biovolume, species richness, Simpson's Index, and chl-a, where each of these responses decreased with increasing velocity. The strength of this predictor across multiple metrics strongly suggests that velocity is important particularly during higher flow periods of the summer. Other explanatory variables during this period were not highly useful in explaining the observed variation.

During the fall, the most important explanatory variable for biovolume and abundance was Flow Daily SD, which is a measure of flow variability. It should be noted that we also considered models using the coefficient of variation and the standard deviation of mean daily flow and the results were similar (data not presented). This suggests that as flow variability increases, there is a subsequent decrease in algal productivity. This is similar to the decrease in cell biovolume observed with increased flow variability during the winter. Velocity was also an important explanatory variable. The data suggests that as velocity increases, the percentage of the community derived from the reservoir decreases, the species richness decreases, and the algal diversity also decreases. The models also suggest that as nutrient addition increased, there was a subsequent increase in chl-a. Although we have modelled upstream nutrient addition, our results are preliminary as we have arbitrarily chosen to use nutrient loadings from four months prior to sampling. It is probable that nutrient addition is affecting downstream productivity, either through import of increased algal cells (more likely) or through direct transport of nutrients (less likely), our models are not sensitive enough to refute or confirm the specific effects and continued investigation is needed.

When considering all time periods and metrics, velocity appears to be a predominant factor. The next most important factor observed, across all metrics seems to be related to flow variability. This result in particular suggests that a direct link between productivity and operations may exist. However, since this is the first attempt to explicitly test the management questions through modelling, results are considered preliminary and further analysis with additional years of data is needed to better understand how flow variability and operations may affect periphyton productivity.

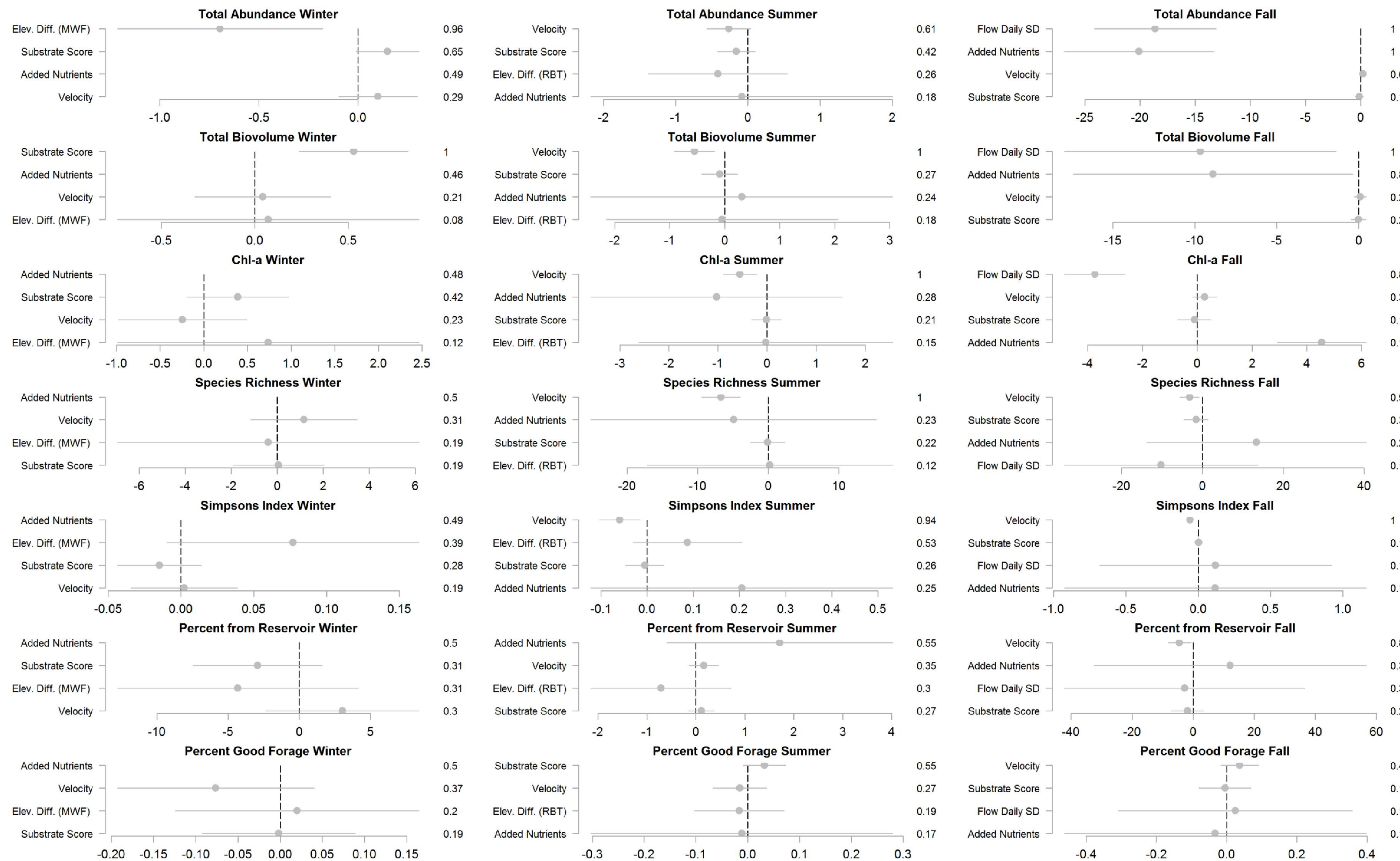


Figure 3-17: Mean coefficients and their 95% confidence limits of standardized explanatory variables of periphyton production in LCR (2009 – 2014). Coefficients are standardized to allow direct comparisons of the direction and size of effects, noting that variables with confidence limits that encompass 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.

3.3.4 Seasonal Effects on Periphyton Production

The seasonal effects on periphyton growth detected by statistical modelling reported in the preceding section are supported by detailed investigation of shifts in taxa, and are reported in this section.

Despite the moderate and stable production in LCR from year to year, there were substantial differences in the composition, abundance and biomass of periphyton observed between the three seasonal deployments in LCR. In all seasons, the highest species richness was observed at the shallowest samplers that remained permanently submerged. Variable flows in the fall temporarily exposed some of the shallow samplers and possibly reduced periphyton production (**Figure 3-18**). In contrast, shallow permanently submerged samplers had more light and lower velocities (less shear) than the deeper sampler positions and were highly productive and diverse. Many of the deep samplers also had high diversity but with increased numbers of tightly attached or stalked diatoms. Species richness was generally low in the fall at the deep sites and high in the summer at the shallow sites, while winter was variable. 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 is far higher than the diversity observed in MCR, indicating that LCR has better growing conditions.

The summer period includes freshet. Summer periphyton production across all years was lower than the other sample periods and did not show a strong pattern of growth along the depth gradient (**Figure 3-18**). The summer period of flood years such as 2011 and 2012 included very high flows that apparently limited periphyton production. The highest biovolume and diversity occurred on the shallow samplers (always submerged) and remained stable with increasing depth, while chl-a increased slightly with depth, reflecting a shift in taxa. LCR production metrics for biovolume and chl-a were correlated (R=0.80), as expected.

Across all years, fall periphyton biovolume was higher than summer production. Samples showed decreasing biovolume with depth after MS, but chl-a peaking at MD (**Figure 3-18**), These results suggest variable contributions of taxa including filamentous green algae, and photosynthetic bacteria, most likely mediated by decreased available light and increased velocity. The shallow sites showed lower productivity and diversity in fall, probably because of periodic dewatering during the fall fluctuating flows. 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 7.5% at deep sites (dead diatom biovolume decreased from 8.5% to 5.3%).

Winter production was measured in 2013 and 2014. The biovolume results were affected by the exceptionally high 2013 production that included prolific growth of Didymo (**Figure 3-18**), while winter chl-a was intermediate between summer and fall, as one might expect. Production was similar at all depths with a possible small increase from shallow to deep. Stable winter flows benefitted Didymo growth and apparently benefitted other components of the periphyton community as well. The abundance of dead diatom frustules was also slightly higher at 7.9% in the winter, compared to 6.0% in the fall and 6.2% in summer, 2014.

Winter 2013 conditions, including high flows (~2000 m^3/s) in January and a substantial drop on February 9th to ~800 m^3/s low stable flows allowed thick periphyton and dead frustules to remain on the substrates. In winter 2014, flows started off around 1,500 m^3/s and gradually tapered downward to ~750 m^3/s. This subtle change in flows between the two winters seems unlikely to cause the substantial drop in periphyton and Didymo growth seen in 2014 samples. Other factors such as water temperature may induce its large mats to form. Didymo mats persist until they were dislodged by rising freshet flows in late April. The winter season is clearly unique with variable periphyton production and a different community structure, including proportionately fewer diatoms and more low light tolerant cyanobacteria than most summer and fall samplers, all resulting in lower forage quality for benthic invertebrates.

Reported abundance and biovolume and chl-a consider live periphyton. Only ash-free dry weight (AFDW or volatile solids) includes all live and dead organic material. Like other metrics, AFDW analyses showed winter 2013 was significantly more productive than winter 2014 and more productive than any other sampled season (**Figure 3-19**). This was caused by very abundant growth of Didymo mats, particularly at S4 and S7 in 2013, but that growth moderated in 2014. Very little periphyton grew on or under the Didymo mats, causing a reduction in metrics including chl-a, autotrophic index, species richness and diversity.

Summer seasons with the freshet flow periods were consistently lowest for AFDW (**Figure 3-19**). This metric is also affected by the number of benthic invertebrates present in the samples. For example, the fall 2014 AFDW results at S4 and S5 were more affected by caddis fly biomass than by periphyton biomass.

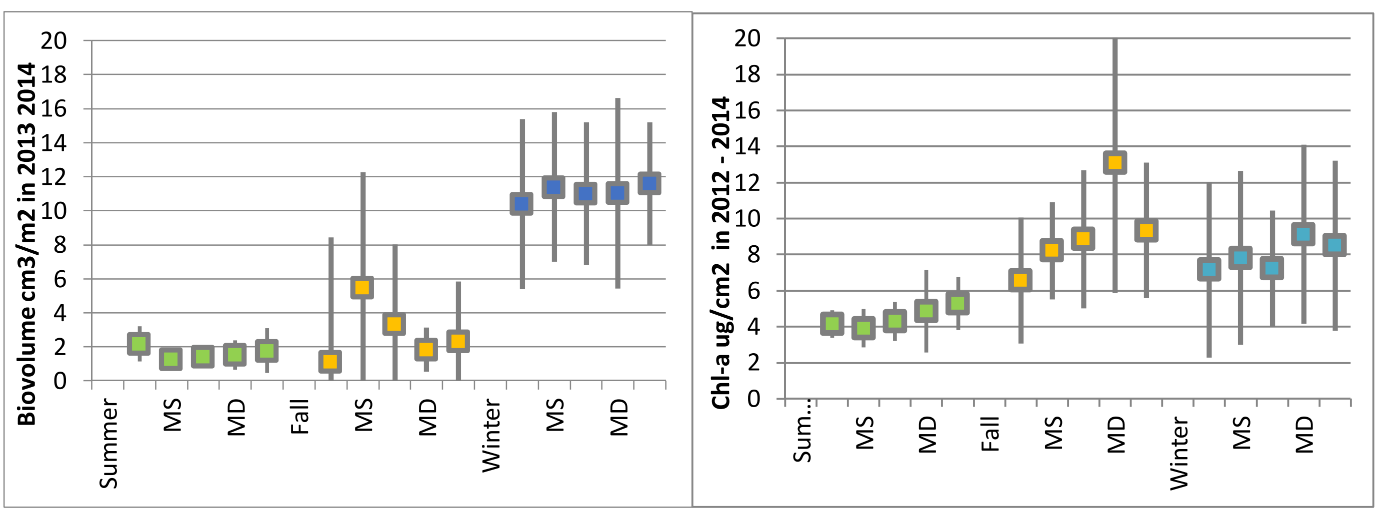


Figure 3-18: Mean Periphyton Biovolume (cm^3/m^2) and Chlorophyll-a ($\mu g/cm^2$) ± SD in Summer, Fall and Winter in 2012-2014, over the Range of Sampled Depths. Depth labels are: S=shallow, MS=moderately shallow, M=mid, MD=moderately deep, D=deep

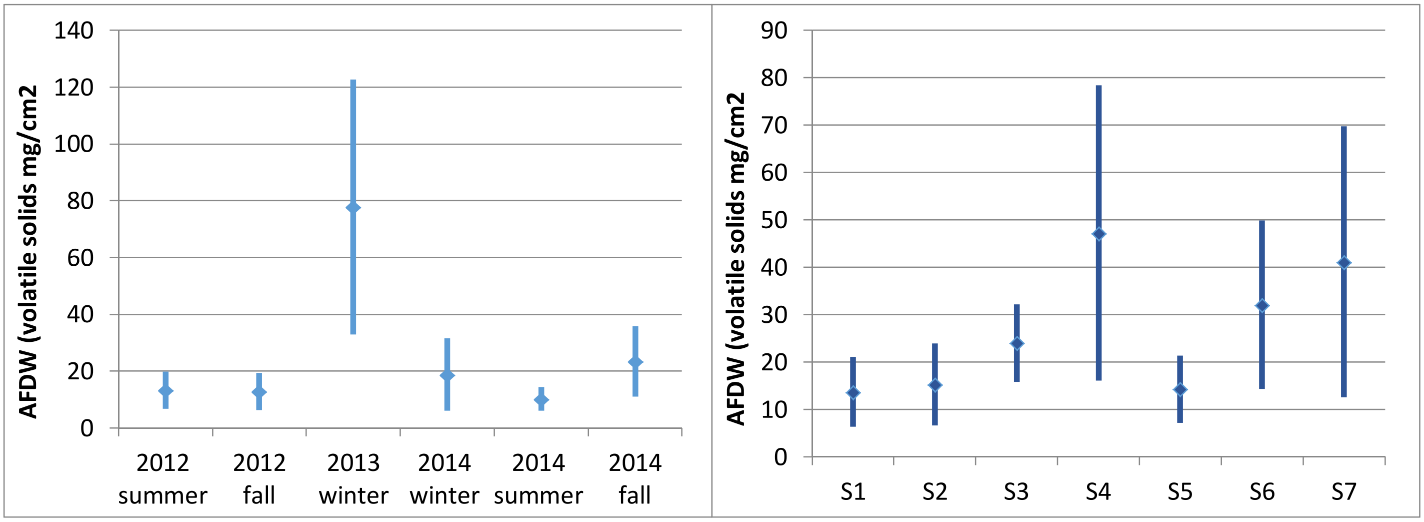


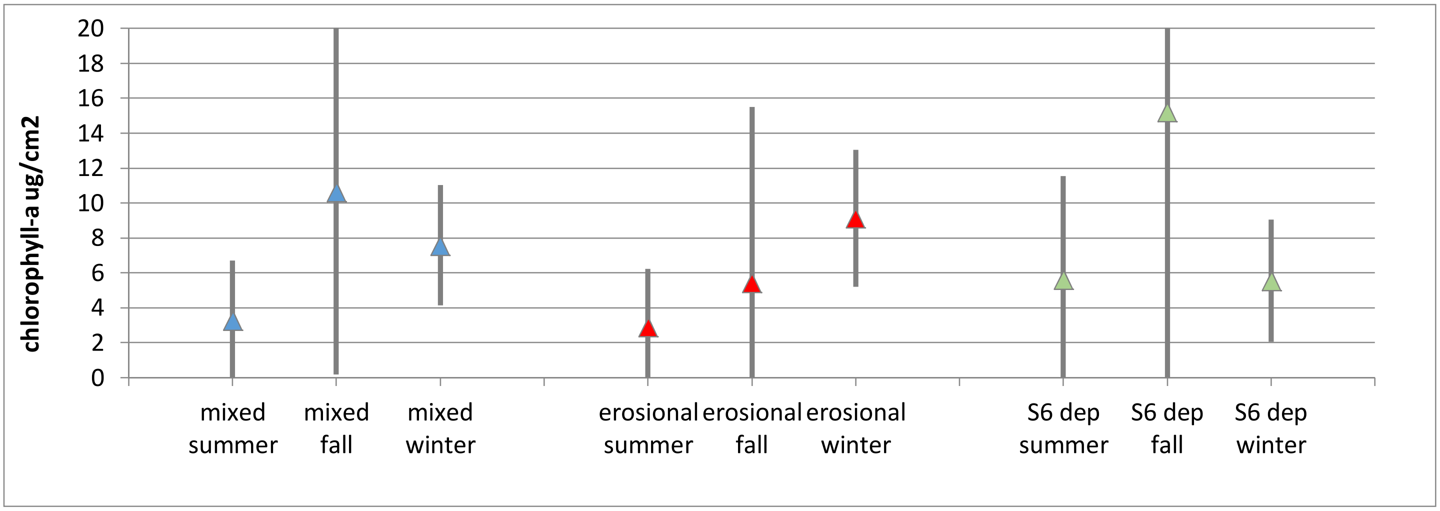
Figure 3-19: Mean Ash-free dry weight (mg/cm2) ± SD by Season and by Site in 2012-2014

Chlorophyll-a and AFDW provide complementary information that 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, where higher values indicate a greater proportion of non-photosynthetic organisms (Biggs and Close 1989; APHA 1995; Biggs and Kilroy 2000; Yamada and Nakamura 2002; Runion 2011). In all seasons, greater autotrophic production occurred on the moderate depth samplers, particularly at erosional sites. The highest heterotrophic production occurred on deep and mid-deep samplers, particularly at depositional and mixed sites. These results imply that proportionately more photosynthetic production occurred at erosional sites, while more decomposition occurred at mixed and depositional sites. Additionally, the winter mid–depth samples with the greatest Didymo and AFDW (3.15 ± 1.75 mg/cm^2) had the lowest chl-a (7.1 ± 3.4 µg/cm^2) and AI values (1410 ± 1380).

3.3.5 Influence of Site Type and Substrate Size

Although not statistically analyzed, there appeared to be a relationship between specific types of periphyton and sites with varying substrate size. Models treated site as a random effect, since management questions tend to focus on larger scale processes. As a result, site level effects were investigated using descriptive methods rather than explicit testing. 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 position of the end of the varial zone with periodic exposure, and the beginning of the permanently wetted substrates, similar to banding patterns 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 m depths (MS transects). As a result, 2014 fall samples had the highest average autotrophic index values averaging 3470 while winter samples had the lowest at 1770 (data not shown). 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 sites dominate in LCR and they had very high winter production, high fall productivity and moderate summer productivity (**Figure 3-20**). Depositional sites are less common in LCR; Site 6 was productive in the summer and fall seasons but was less so in the winter. 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. The mixed sites behaved similarly to the erosional sites, where summer production was low, followed by improved periphyton production in the fall and winter. Viable Didymo mats were rarely encountered on the depositional substrates in any season, rather, they occurred on cobble substrates that experienced low velocity flows.



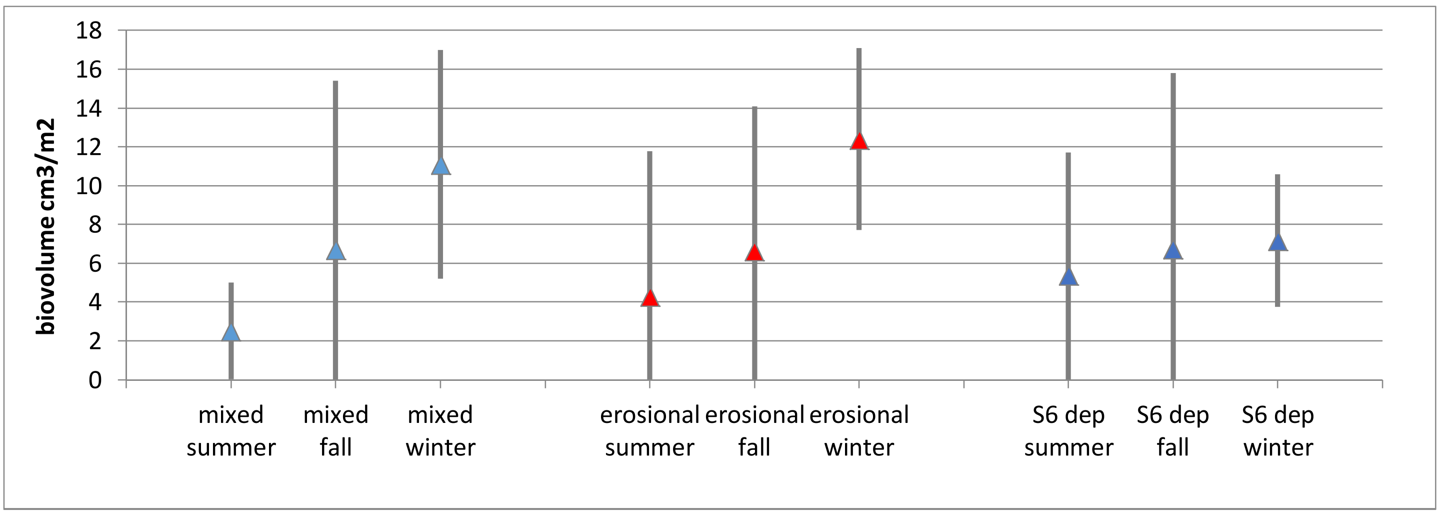


Figure 3-20: Mean Periphyton Chlorophyll-a ($\mu g/cm^2$) and Biovolume (cm^3/m^2) ± SD in Summer, Fall and Winter 2008 - 2014, for All Sites Combined, Erosional Sites S1,S2,S7 and Depositional Site S6.

When all site types are considered together, the summer deployment with freshet and record flows had the lowest production, followed by the fall with fluctuating flows, while the winter with its stable low flows produced more chl-a than the longer winter deployment. Periphyton production in erosional and depositional sites was similar in the summer but depositional sites had more chl-a in the fall, while erosional sites had more productivity in the winter. These results agree well with the statistical models that consistently identified flows, flow variability and velocity as the most important factors influencing periphyton production in LCR.

3.3.6 Influence of Reservoir Phytoplankton on LCR Periphyton

Drifting Arrow Lakes Reservoir phytoplankton settle onto the LCR periphyton, increasing periphyton diversity and standing crop. Algae density in the drift was low in April/May samples, while the highest densities occurred in the August/September period (Digital Appendix). The late summer/fall drift composition was also more diverse, with more soft-bodied green and flagellated algae. The fall periods were affected by deeper mixing of the reservoir water columns and possibly by the expected arrival of ALR fertilizer nutrients and the algae those nutrients stimulated (Schindler *et al.* 2006). Over 60% to 95% of the drift diatoms and filamentous cyanobacteria were exclusively lake forms. Some of these reservoir genera will gradually die off in the periphyton such as *Asterionella* and *Fragilaria*, while other genera such as *Diatoma spp*. and *Tabellaria spp.* are known to persist in the periphyton of downstream rivers that have stable flows (Bonnett *et al.* 2009).

Drift in the Kootenay River usually had more algae cells and chl-a than LCR drift and would benefit LCR periphyton production below the confluence. However, drift chl-a samples collected in August and October 2014 showed identical results of 0.65 and 2.0 µg/L, respectively. Many of the large reservoir diatoms remained suspended over many kilometers of river flow.

There was significant seasonal and inter-annual variation in the contribution made by reservoir phytoplankton to the periphyton in LCR. Over the years of sampling, an average of 22±8% of the periphyton seen on periphyton samplers in the summer and fall was attributable to reservoir phytoplankton, while only 2 ±1.4% was phytoplankton in the winter samples. Donations from the reservoir are therefore not responsible for the significant winter periphyton growth. Within the stable summer and fall seasons, however, reservoir periphyton contributions ranged from 11 ± .2% (2014) to 42.5 ± 12% (2010). These are significant contributions.

Plankton hauls also contained numerous zooplankton donated to LCR by ALR and Kootenay Lake, particularly in August. They were dominated by copepods, *Daphnia, Keratella* and *Kellicotia* from ALR, and copepods, *Simulidae*, *Bosmina* and *Keratella* from Kootenay flows. These zooplankters can be utilized by fish. None of the plankton hauls collected mussel veligers, and no invasive mussels were seen during examination of piers and pilings in the LCR to date.

3.3.7 Value of Periphyton to LCR Food Chain

Many components of the periphyton are good food for benthic invertebrates that are in turn key diet items for fish. The diversity of erosional, depositional and mixed sites in LCR provides a range of feeding opportunities for benthic invertebrates including grazers, collector/gatherers, scrapers and omnivores. Additionally, drifting algae from reservoir releases also provides food.

Overall, the periphyton forage quality ranged from good to poor. Most periphyton diatoms provide good forage. Large filamentous green species may not be directly edible, but they create microhabitats that can harbour key food organisms. Unlike Didymo filaments, moderate growths of green filamentous algae are beneficial to LCR productivity. The percentage abundance of good forage taxa ranged from 7% to 51%, averaging 24±10% overall, and was seasonally stable. Erosional sites averaged 24±11%, mixed sites averaged 25±8, while depositional Site 6 averaged only 17±6%, supporting our observations that benthic invertebrate production was greatest in the cobble/gravel substrates that are typical in LCR. The proliferation of the inedible Didymo during stable winter flows reduces edible periphyton production on the cobble substrates at mid-depths.

3.4 Benthic Macroinvertebrates

During the three sampling sessions in 2014, 90% of rock baskets were recovered (**Table 3-7**). Most of the loss occurred at shallow depths and was a result of human tampering (i.e. samplers pulled out of water).

Table 3-7: Rock Basket Recovery by Season in 2014. *Fractions indicate the number of substrates recovered over the number of substrates deployed.*

Season	2014
Winter	32/35
Summer	33/35
Fall	30/35

3.4.1 Brief Summary of Benthic Invertebrate Sampling

In 2014, LCR had an abundant and diverse community of benthic macroinvertebrates. Rock basket sampling resulted in the collection of 28 different taxa in both the winter and summer and 32 taxa during the fall (Digital Appendix: max species richness). The relative abundance of benthic invertebrates was assessed at the family or genus taxonomic level, while relative biomass was grouped according to either class or order since biomass was only determined for larger groups.

The 2014 benthic invertebrate data varied by season. The highest mean abundance (#/basket) ± SD occurred in the summer with 7973±5828 organisms per basket, followed by fall and winter with 6684±4625 and 3783±3438, respectively (**Figure 3-21**). Biomass also trended differently across season. Winter had the highest biomass, followed by fall and summer was substantially lower (**Figure 3-22**). Although the benthic invertebrate data typically varies between years, the 2014 data fell within the range of previous sampling periods (**Figures 3-21** and **3-22**).

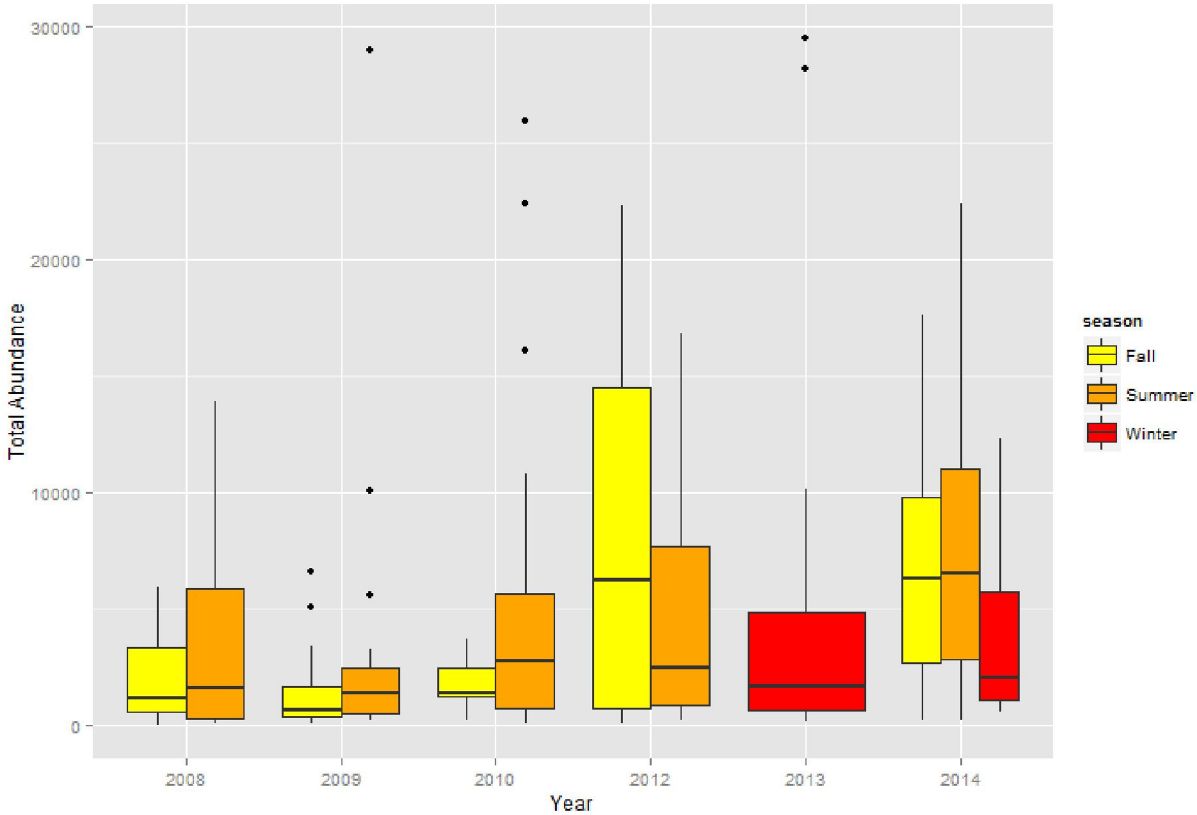


Figure 3-21: Total Abundance of Benthic Invertebrates grouped by Season and Year.

Mean species richness numbers were very similar across the three seasons ranging from 16.03 ± 4.9 in the winter, to 18.82 ± 4.3 in the summer and 18.24 ± 4.5 in the fall (Digital Appendix: mean species richness). Dominant taxa in the summer and fall included Hydropsychidae (net-spinning caddisflies), *Tvetenia (*non-biting midge; Chironomidae*)* and *Parachironomus* (non-biting midge; Chironomidae). In the winter, a different suite of organisms dominated. Three taxa including Simuliidae (black fly), *Simulium* (black fly) and *Orthocladius* (non-biting midge; Chironomidae) comprised approximately 85% of the samples. The dominant taxa sampled during each season were very similar to those documented in 2013. 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 56.3 and 73.4 percent of the relative biomass. In contrast, Trichoptera during the winter comprised less than 1%. Diptera (86.2%) and Ephemeroptera (9.1%) maintained the greatest relative biomass in the winter.

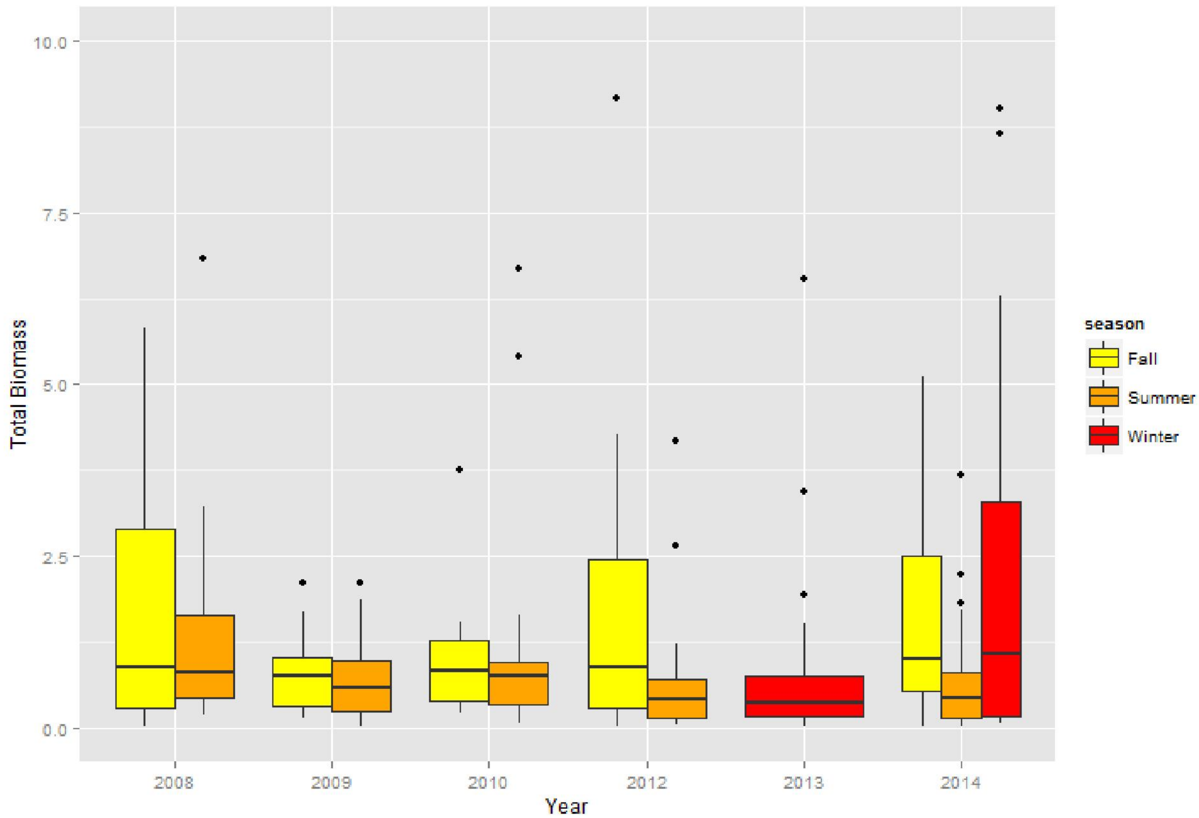


Figure 3-22: Total Biomass of Benthic Invertebrates grouped by Season and Year.

The mean Hilsenhoff Biotic Index (HBI) was 5.7, 4.6 and 5.3 for winter, summer and fall, respectively. The HBI ranges from 0-10; taxa with a zero value are extremely intolerant of pollution, taxa with scores of 2 – 9 have varying degrees of tolerance, while a score of 10 indicates a high ability to withstand pollution. Pollution sensitive species are typically higher quality food for fish and their presence is indicative of a healthier system.

In 2013, we reported a modest trend of increased abundance of benthic invertebrates with increased depth in the winter only. This trend was not apparent in the summer or fall datasets that extended across multiple years. Abundance data from 2014 corroborated that of previous years. We suspect that the stable, low flow conditions of the winter drive the benthic invertebrate community to deeper sites, while the higher, more variable flows during the summer and fall sampling periods make the deepest sites uninhabitable for many taxa. In 2014, the highest benthic invertebrate abundance during both the summer and fall was documented at the moderately shallow sites.

3.4.2 Benthic Invertebrate Community Groups

Community analyses of the 2008 – 2014 invertebrate data was also completed at the genus level. NMDS reduced the genus level community compositions to 2 dimensions/axes. The stress index was 0.22, which indicates the two NMDS axes partially explain the invertebrate community composition. A permutational MANOVA indicated that invertebrate community compositions exhibited significant differences when grouped by year ($F=16.2$, $p=0.001$), season ($F=23.2$, $p=0.001$), depth ($F=1.76$, $p=0.002$), and site ($F=3.76$, $p=0.001$). However, season explained the most variation ($R^2=0.12$) in invertebrate community compositions compared to year ($R^2=0.05$), depth ($R^2=0.02$), and site ($R^2=0.07$).The stronger separation of invertebrate community compositions in terms of season is also seen in **Figure 3-23**.

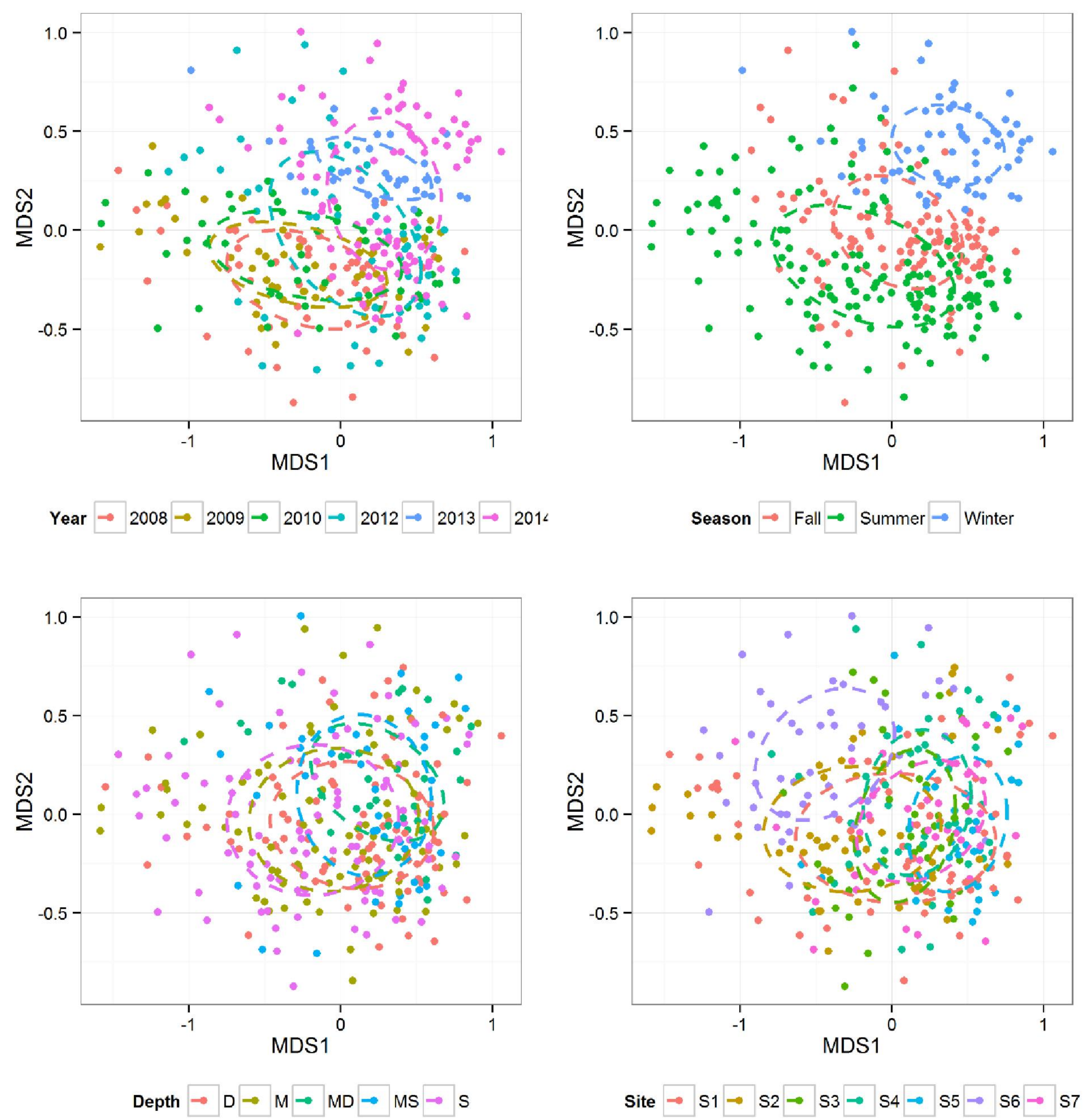


Figure 3-23: NMDS of Invertebrate Genus Level Abundance grouped by Year, Season, Depth, and Site for all Data between 2008 – 2014. The NMDS used a Bray-Curtis dissimilarity index and had a stress index of 0.22. Ellipses are calculated based on 95% CI of the NMDS scores for each group.

3.4.3 Benthic Invertebrate Production Models

Model averaging of key benthic invertebrate responses included total abundance, total biomass, species richness, Simpson's index, percent EPT, percent Chiron, Hilsenhoff Biotic Index, and available biomass of good food for fish (sum of EPT and Dipterans). 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 2008. For each response, the following explanatory variables were used: added nutrients, elev. diff. (MWF) (winter only), elev. diff. (RBT) (summer only), flow daily SD (fall only), velocity, and substrate score. A detailed description of each explanatory variable is included in the methods (Section 2.5.5).

The number of plausible models (those with an AICc<3.0) ranged from 2 to 8 across all seasons and the total number of models considered was 128 for each season (**Appendix A-5**).

Winter: Elevation Diff (MWF), a measure of flow variability, was the most important predictor of percent EPT, Simpson's Index, and species richness during the winter, suggesting that operations may have a direct effect on benthic invertebrate productivity (**Figure 3-24**). The negative coefficient means that the percent EPT, Simpson's Index and species richness of benthic invertebrates decreased with an increasing water elevation difference between MWF spawning (Jan 1-21) and MWF incubation (Jan 21-Mar 31). These are the first results of this study that suggests that the management of the MWF flow period may have an effect on the benthic invertebrate community. The data is preliminary and additional sampling during the winter is needed to confirm the relationship, but the strength of this predictor across multiple metrics strongly suggests that flow variability is important during the winter. Velocity was also an important explanatory variable for total abundance, total biomass, and good food, where each of these responses increased with increasing velocity. This suggests that during the winter when flows are typically low, higher velocity sites exhibit a greater productivity. Substrate score was the other metric that was as important predictor of the Hilsenhoff Biotic Index. This metric decreased with increasing substrate size. This is expected given that HBI tolerant taxa (e.g., Chironomidae) are more prevalent in areas of finer substrates.

Summer: In the summer, substrate score was the most important predictor of total abundance, Hilsenhoff Biotic Index, percent EPT and good fish food (**Figure 3-24**). The RVI values for this metric ranged from 0.79 to 0.92 indicating that substrate score was a useful explanatory variable. Total abundance, percent EPT and good food increased with increased substrate size, while similar to the winter Hilsenhoff Biotic Index decreased with increased substrate size. The summer models also suggest that as nutrient addition increased, there was a subsequent increase in the Simpson's Index and percent Chironomids. Although we have modelled upstream nutrient addition, our results are preliminary as we have arbitrarily chosen to use nutrient loadings from four months prior to sampling. It appears that it is probable that nutrient addition is affecting downstream productivity, either through import of increased algal cells (more likely) or through direct transport of nutrients (less likely), but our models are not sensitive enough to refute or confirm the specific effects. Continued investigation is needed. Velocity was the most important predictor of species richness. As velocity increased, species richness decreased. This relationship also held true in the winter and fall. It indicates that higher velocity sites are limited to certain groups such as EPT that are capable of withstanding higher velocities. Elevation diff. (RBT), a measure of flow variability, was the most important predictor of total biomass in the summer. The RVI value for this metric was 0.73 indicating that elevation diff (RBT) is only moderately useful in explaining the observed trends. Nevertheless, the trend is similar to what was observed in the winter, increased flow variability results in a subsequent reduction of biomass.

Fall: During the fall, velocity was the most important predictor for four of the six response variables (good fish food, HBI, percent EPT, Simpson's Index, species richness, and total abundance) (**Figure 3-24**). The RVI values for this metric ranged from 0.62 – 1.0, indicating that velocity was generally a useful explanatory variable. As observed in other seasons, good fish food, percent EPT, and total abundance increased with increased velocity, while HBI, Simpson's Index and species richness decreased with increased velocity. All other models tended to explain less variation (had low R^2) and had explanatory variables that were not useful in explaining the observed trends (RVI's < 0.6).

When considering all time periods and metrics, velocity appeared to be the dominant factor. Variability in flow was important during the winter and to a lesser extent during the summer and fall. This result in particular suggests that a direct link between productivity and operations may exist. However, since this is the first attempt to explicitly test the management questions through modelling, results are considered preliminary and further analysis with additional years of data is needed to better understand how flow variability and operations may affect benthic invertebrate productivity.

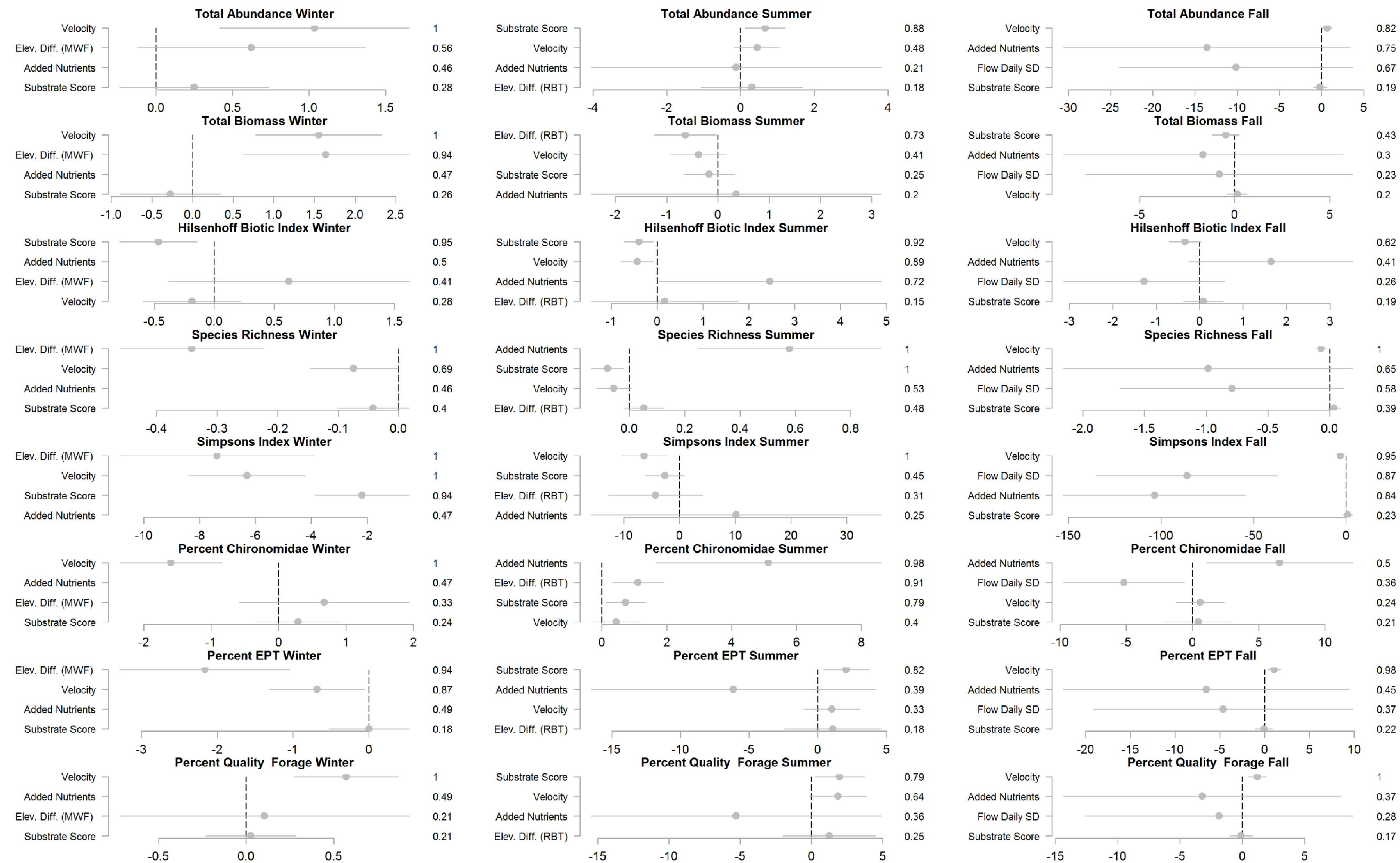


Figure 3-24: Mean coefficients and their 95% confidence limits of standardized explanatory variables of benthic invertebrate production in LCR (2009 – 2014). Coefficients are standardized to allow direct comparisons of the direction and size of effects, noting that variables with confidence limits that 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.

3.4.4 Fish Food

The benthic invertebrate samplers have been successful at capturing organisms that are representative of diet data for both MWF and RBT (Golder Associates Ltd. 2009). Continued implementation of fish flows and the potential effects on the availability of fish food organisms was assessed as part of the benthic invertebrate modelling. Three response variables including the percent biomass of Chironomidae, percent biomass of EPT and percent quality forage (% EPT and Diptera Biomass) were incorporated to specifically test the effect of flow management on food for fish (**Figure 3-24**).

The model results for % EPT and % forage quality were very similar, suggesting that the effects of food for fish may be driven more by EPT biomass than that of Dipterans, the next most important forage group. The data suggests that the quantity of good forage is positively associated with substrate size and velocity during the summer and fall. During the winter % forage quality was positively associated with velocity but % EPT was negatively associated. We suspect that the negative association during the winter is due to lower relative abundances of EPT taxa, as Diptera was much more abundant during the winter. Velocity was the most important explanatory variable driving % forage quality in the winter and fall, while substrate score appeared to be more important during the summer.

Interestingly, the % Chironomids within LCR appear to be positively associated with nutrient additions in upstream areas, at least during the summer and fall periods. Also, velocity appears to have a more negative influence on Chironomids than EPT taxa, particularly during the more stable winter periods. However, this may be related to site level effects from areas with a greater predominance of fine substrates than actually due to a direct effect of lower velocities. Finally, the effects of flow regulation appear to have the greatest effect on % Chironomids when compared to EPT taxa, where the % Chironomids is negatively associated with daily variability in flows during the FFF and the elevation differences during the MWF flow period. However, during the RBT flow period, the % Chironomids in the community appears to increase as the cumulative drop in water elevation increases.

4.0 DISCUSSION

4.1 Water Temperature

Water temperatures varied seasonally, ranging from approximately 3 to 19°C and were 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, followed by upstream reservoir temperature, and reservoir elevation. The data suggest that flow does influence water temperatures to some extent, but the specific effects were variable and depended on the flow period in question. This is the first year that separate statistical models for each flow period were generated, allowing us to eliminate interactions and to directly test the effects of flow within each flow period. The models determined that during the MWF flow period, riverine temperature is more dependent on reservoir temperature than air temperature, whereas during other flow periods, air temperature is a more important factor. 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.

We therefore continue to tentatively 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. Although flow has the potential to affect river temperature, the data suggests that other parameters such as air temperature, reservoir temperature, and to a lesser extent reservoir elevation and stratification are probably more important determinants of river temperature than flow or operating regimes during specific flow periods. From a physical perspective, this is logical because a substantial amount of energy would be required to either heat or cool the normal volumes of water within the LCR during any given flow period and simply adjusting how much water flows is not likely to provide or remove sufficient energy to the system to result in large shifts in water temperature due to flow regulation.

4.2 River Flows

The 2014 freshet peaked on July 8th, with approximately 3,677.9 m^3/s recorded at the Birchbank Gauging Station. For comparison, the maximum mean daily river flows recorded in previous years of this study (2011 – 2013) were 4,155.4 m^3/s on July 9th, 6,043.1 m^3/s on July 21th and 4,434.4 m^3/s on July 5th, respectively. The freshet peak occurred after 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).

Historical water elevation data was not available so a predicted data set was used to estimate historical water elevations. Since channel morphology has not significantly changed since 1984, a reasonably accurate prediction is possible because river elevation is a function of channel morphology for the most part. In wider channels, larger changes in flow are required to obtain the same changes in elevation when compared to narrow channels. For this reason, each elevation station was modelled independently to address site-specific channel morphological effects in the analysis.

The modeling data indicate that both of the post-implementation (1995 – 2007) and continued (2007 – 2014) 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 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. Similar to 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. 2014 was the 7th year of this study and there was sufficient data to explore additional analyses beyond simple data summaries. Based on this first modeling exercise and our previous experience with LCR water quality, we continue to tentatively accept the management hypotheses HO_{3phy}, HO_{3Aphy}, and HO_{3Bphy} and assume that MWF, RBT or FF flows have no effect on the water quality of LCR.

Conductivity, TDS, alkalinity and pH were sampled to address electrochemistry, while the sampled biologically active nutrients included nitrate, ammonia, and ortho-phosphate (SRP). Nutrients occurring as organic particulates require bacterial digestion before they are returned to a biologically active form, and of these, total phosphorus, TKN and total nitrogen were analyzed. Currently, sampling occurs once per season, whereas in the early years of the study water quality sampling occurred only between June and October. The limited sampling is difficult to directly test specific effects of flow because chemistry changes on a daily, weekly, and monthly basis. This means that addressing the water quality hypotheses (HO_{3phy} HO_{3Aphy} HO_{3Bphy}) remains challenging.

LCR water quality sampling is intended to provide an understanding of river water chemistry, and its influence on productivity. Overall LCR water chemistry is set by HLK and BRD dam releases which together account for 98% of flows. Most water quality parameters varied between years and distinct seasonal patterns due to flow (e.g., freshet) were observed. Some water quality parameters did not appear to vary with flow. For example, both the Kootenay and Columbia systems showed stable pH throughout the study, even during the record 2012 freshet (Larratt *et al.* 2013). Subtle flow changes such as fish flows should therefore have a very minor influence on parameters that do not directly vary with flow such pH.

For those parameters that did vary with flow, the effects were frequently in proportion to the flow event. For example, the sediment carrying capacity of flowing water is proportional to its velocity (Hei *et al.* 2009; Giller and Malmqvist 1998; Hem 1985). For this reason, both turbidity and suspended solids were low during winter and fall flows on the regulated Columbia and Kootenay Rivers, with spikes during freshet. Further, a direct, inverse relationship was evident between flows in LCR and electrochemistry. Higher flows in the spring and early summer diluted dissolved solids and resulted in lower conductivity readings. After freshet, when lower summer and fall flows resulted in less dilution, these parameters generally increased. During these lower flow periods with lower water elevations, there is presumably a greater contribution from groundwater into the base flows which would increase electrochemistry (Peterson and Connelly 2001; Tolan *et al.* 2009; Golder 2010). Therefore, periods where flows are manipulated for fish should act on electrochemical parameters, but the influence of fish flows would be very small compared to larger flow events.

An effect of flow on total nutrients was also observed and is likely the result of suspended organic detritus, sediment, phytoplankton and dislodged periphyton as measured in T-P, TKN and T-N. For example, total phosphorus concentrations spiked in 2014 immediately following a storm because more particulates were scoured into suspension. Similarly, organic nitrogen (TKN) increased during high flows by 25% in LCR and by 46% in Kootenay River. If the observed response of LCR to freshets and storm flows is broadly applicable, then moderately increased flows may improve the delivery of nutrients to periphyton. However, very high flows can increase scour and lower productivity while increasing total (biologically unavailable) nutrients. The statistical models, although interpreted with caution, also suggest that high daily flow variability has a direct influence on T-P in LCR, likely through the scour mechanism discussed above. Variation in daily flow and operations during the MWF and RBT flow periods were also plausible predictors, with variation in daily flow having the greatest importance on T-P. This suggests that high fluctuations in flow have a direct effect on total nutrients in the LCR and that increased flow variability likely increases total nutrient concentrations, similar to the observed effect of a storm event. The influence of flows on total nutrient concentrations must also consider upstream conditions in Arrow Lakes and Kootenay reservoirs, particularly for total phosphorus, because algae cells exported seasonally to LCR will increase T-P.

In any river system, there are numerous correlated influences on the biologically active (inorganic dissolved) nutrient concentrations. Dissolved nutrient concentrations in LCR are affected by factors including the limnology and nutrient status of Arrow Lakes Reservoir (Hatfield, 2008) and Kootenay Lake along with their respective fertilization programs, the numerous outfalls that exist on LCR, tributary and groundwater inputs. Unlike flow-induced scour influencing total nutrient concentrations, anthropogenic dissolved phosphorus sources are independent of flows and are therefore more likely to be diluted by higher flows. Throughout this study, ortho-phosphate (or SRP) rarely exceeded the detection limit of 0.01 mg/L in LCR samples. These results are all lower than the historic range recorded for Birchbank, and continue to follow a declining trend in LCR over the years (Holmes and Pommen 1999). During high freshet years, more inorganic nitrogen was observed in LCR than in years with lower peak flows (Scofield *et al.* 2011; Larratt *et al.* 2013). Within each year, seasonal effects occurred where summer nitrate concentrations were lower than the other seasons. Modelling of nutrients showed that nitrate+nitrite increases with greater daily flow variability.

Previously, we had preliminarily accepted the management hypotheses and had assumed the fish flows had no effect on LCR water quality. Our reasoning was that the influence of fish flows on water quality was more subtle compared to the effects of freshet for instance. The modelling to date supports the theory that operations during fish flow periods are much less important than daily variation in flow on overall water quality, although daily operations may be a factor in nutrient cycling within the system. Interestingly, Arrow Lakes Reservoir water quality parameters were also included as predictor variables, but the models did not identify them as important predictors of LCR water quality. However, this may be the result of how we incorporated the data as a predictor because many assumptions were made rather than because there is a lack of relationship between upstream and downstream conditions.

4.4 Periphyton Monitoring

The ecological monitoring management question HO_{2eco} addresses the effect of implementation of MWF, RBT and FFF on total biomass accrual of periphyton in LCR. To address this management question, periphyton monitoring during 2014 was concentrated in Reach 2 at seven sites and in the areas presumed to be most productive, ranging from the water's edge to 5 - 6m deep. Deeper thalweg areas can exceed 10 m depth and were not sampled because they are extremely difficult to sample, and we speculate that they have very little, if any, light penetration due to water depths.

4.4.1 Periphyton Production and Community Metrics in the LCR

The LCR periphyton community is productive, diverse and variable[3]. Most production metrics place LCR in the typical to productive range compared to other large rivers (**Table 4-1**). Although there was high variability between seasons and years, the 2008 – 2014 accrual data indicate that peak biomass occurs in 6-7 weeks in the summer, greater than 8 weeks in the fall and in 10 – 12 weeks in the winter.

[3] It is important to note that our artificial substrates may present a bias on results and we estimate the effect to be as large as 50% of the observable productivity on natural substrates (Larratt *et al.* 2013). However, our data also suggest that the observable abundance or biovolume between the Styrofoam artificial substrates (closed cell Styrofoam) and that of stone tiles is similar, meaning it is reasonable to directly compare results from this study to the larger body of river periphyton research utilizing stone tiles which are most representative of natural substrates.

Table 4-1: Summary of typical LCR periphyton metrics from 2008 to 2014, with comparisons to oligotrophic, typical, and productive large rivers and MCR

Metric	Oligo-trophic or stressed	Typical large rivers	Eutrophic or productive	MCR	LCR
Number of taxa (live & dead)	<20 – 40	25 - 60	variable	5 - 52	8 - 60
Chlorophyll-a μg/cm^2	<2	2 - 5	>5 – 10 (30+)	0.04 – 4.1	0.04 – 43
Algae density cells/cm^2	<0.2 x10^6	1 - 4 x10^6	>10 x10^6	<0.02 – 1.5 x10^6	0.03 – 3.9 x10^6
Algae biovolume cm^3/m^2	<0.5	0.5 – 5	20 - 80	0.03 - 10	0.1 – 25
Diatom density frustules/cm^2	<0.15 x10^6	1 - 2 x10^6	>20 x10^6	<0.01 – 0.6 x10^6	0.25 – 2.3 x10^6
Biomass –AFDW mg/cm^2	<0.5	0.5 - 2	>3	0.12 – 4.8	0.35 – 7.1
Biomass –dry wt mg/cm^2	<1	1 – 5	>10	0.7 – 80	2.5 - 101
Organic matter (% of dry wt)		4 – 7%		1 – 10%	0.74 - 13 %
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	0.01 – 0.61

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, along with variable contributions made by soft-bodied algae such as filamentous greens and cyanobacteria. Species richness was lowest in the fall, particularly at the shallow sites and at deep sites where light penetration was lower. It was highest in the summer at shallow and mid-shallow sites because higher flow conditions favoured a broader range of taxa. These substrates also experienced variable scour as flows changed. Over time, 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. This banding pattern was similar to banding patterns observed in MCR and was most observable during the fall.

Upstream reservoirs donated 22 ± 8% of the diatom community during all periods except winter months when only 2 ± 1% was reservoir derived, similar to other river systems immediately downstream of a reservoir (Truelson and Warrington, 1994). Drift during the fall had the highest density and diversity of algae because the fall period is affected by deeper mixing of the reservoir water columns and by the expected arrival of ALR fertilizer nutrients mostly via algal cells (Schindler *et al.* 2009). The arrival of this surge in drift algae occurs at a time when flows are moderate, allowing more of the lake forms to join the periphyton mat. This effect is supported by the modeling data which show a decrease in higher reservoir derived taxa as velocity increases (Bonnett *et al.* 2009).

Overall, the mean daily flows during the FFF and winter period were similar and stable when compared to the summer. These moderate flows allowed more periphyton growth, whereas the summer periods consistently had the lowest periphyton productivity. The highest flow year (2012) had the lowest overall productivity of the study, particularly in the summer sampling period. Reduced periphyton growth following high flow events is frequently observed in other river systems (Blinn *et al.* 1995, Biggs 1996, Bunn and Arthington 2002) and high velocities and increased shear stress are mechanisms identified by models that result in lower overall productivity. The high flows affect periphyton filamentous green taxa and Didymo more than the diatoms, and are responsible for species / genus level shifts in periphyton community structure observed. This change in community structure occurs because filament mats can be dislodged readily from the stream bed with small increases in velocity, while tightly attached diatoms require increased shear stresses to have the same observable change (Biggs 1996).

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 flows because winter samplers showed higher overall periphyton production than summer 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 relative abundance of filamentous green algae in winter samples but more low light tolerant cyanobacteria, diatoms, and Didymo. Stable LCR winter flows permitted the thickest growth of the invasive Didymo seen in the annual cycle. The alteration of natural flows to a more stable pattern can increase the success of invasive aquatic species (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 (Shelby 2006). LCR meets these requirements and this helps explain the predominance of Didymo observed during the winter sampling sessions.

Generally, larger substrates increased overall productivity, particularly for periphyton biovolume, but this effect was not consistently identified in the statistical models. LCR is dominated by erosional cobble substrates, while depositional areas are comparatively rare. The erosional sites benefitted the most from the low, stable flows, which subsequently reduce velocity. Seasonal and site-level effects on community structure and productivity were observed. For example, Site 6 had more decomposer organisms, particularly in the summer when water temperatures of 10 -18^{o}C favoured their growth (Wetzel 2001). Differences between site types were not as apparent during the summer and fall sampling periods, but production at depositional areas was noticeably lower during the winter when depositing sediment hindered periphyton growth relative to the summer, as identified in the models. Determining the influence of substrate and site-level effect remains challenging due to high annual variability.

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 from reservoir releases. However, high variability in the statistical models reduces their predictive power regarding the effects of flow periods on periphyton productivity and community structure.

4.4.2.1 Spring/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]. For example summer 2012 with record flows had the lowest productivity of any sample period to date.

Although freshet flows overshadow the RBT flows both in scale and apparently in effect on periphyton, we still anticipate a subtle effect of RBT flows, thus we now tentatively reject the null hypothesis HO_{2Beco} that RBT flows do not increase total biomass accrual of periphyton in LCR.

4.4.2.2 Fall Fluctuating Flows

Daily flow variability affected physical parameters such as velocity and shear in the river to some extent. Together these flow-related factors affected fall periphyton production and community structure. 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, with the exception of several shallow sites. Periodic low water cover of shallow substrates along the water's edge likely reduced their fall periphyton production. These 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. However, the effects during the FFF are presumably much smaller than the effects caused by larger-scale flow changes.

Based on the data and modelling conducted thus far, we expect a subtle effect of FFF on periphyton productivity, thus we now tentatively reject the null hypothesis HO_{2Ceco} that fall fluctuating flows do not increase total biomass accrual of periphyton.

4.4.2.3 Winter MWF Flow

Like the other seasons, winter operations likely affect periphyton via changes in velocity or water depth associated with changes in flows. The MWF operational pattern is intended to reduce the difference in water elevation over the winter flow period, and this may reduce the overall impact of ramping and substrate dewatering on periphyton production. Similar to the fall, winter biophysical conditions with stable flows are likely more responsible for determining overall productivity and diversity than the specific effects of the MWF operating regime. The LCR winter periphyton model suggests a positive link between substrate score and total abundance, possibly the result of changing light conditions affecting the highly abundant cyanobacteria. Similarly, the subtle differences between the MWF flows in 2013 and 2014 probably do not account for the drastic change in Didymo densities between those two winters.

[4] Note that increased flows do not always directly translate to increases in velocity, but generally, on a larger scale, as flow increases, velocity also increases.

For these reasons, we continue to tentatively reject hypothesis HO_{2Aeco} that MWF flows do not increase total accrual of periphyton or their biomass.

4.5 Benthic Invertebrate Monitoring

The MWF and RBT flow periods have been implemented on LCR for enough time that resulting shifts in the benthic invertebrate community should have stabilized (Poff and Zimmerman 2010). This study was undertaken after the implementation of flows, and five years of benthic invertebrate data have been collected between 2008 and 2014. Thus, only inferences can be made about the potential effects of the implementation of MWF and RBT flows since data is not available prior to implementation. Given this, our approach has been to understand how flows and other physical conditions affect benthic invertebrate communities and subsequently use inferences to understand changes associated with flow regulation.

4.5.1 Benthic Invertebrate Community Structure and Production

Table 4-2 provides a comparison of benthic invertebrates in different river systems. The benthic community in LCR is remarkably more stable, diverse and productive than that of the MCR. This is apparent when comparing the mean number of invertebrates per sample. The 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 (Poff and Zimmerman 2010). For example, impoundment favours the proliferation of orthoclad chironomids (Munn and Brusven 1991). In LCR, chironomids are top contributors to relative abundance. An increased predominance of filter feeding benthic invertebrates has also been documented in regulated river systems and LCR has high relative abundances of web spinning caddisflies of the family *Hydropsychidae*. Thus, in some aspects, the LCR benthic invertebrate community is typical of a highly modified river system. Coupled with the effects of regulation on the invertebrate community, other variables such as nutrient additions through the fertilization program (Schindler *et al.* 2009), industrial effluents (Celgar), municipal effluents, and invasive species (Didymo) all influence the overall distribution, abundance, and diversity of the 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 (e.g., Bunn and Arthington 2002). Thus, specifically elucidating the effects of flow regulation from other stressors and inherent natural patterns on the benthic community cannot be done with certainty.

Table 4-2: Comparison of Benthic Invertebrate Communities in Different River Systems

River	Average Annual Discharge (m^3/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)	1,997	3575(±2093)	40	0.65	Hydropsychidae (25) Parachironomus (9) Tvetenia discoloripes gr. (7.2) Synorthocladius (5.1)
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.

As in most rivers, invertebrate communities in LCR are distributed differentially across the river channel. Previous modeling data suggested that densities were greatest during periods of stable flow in moderate depth areas. High peak flows in 2012 appeared to reduce benthic diversity, and to a lesser extent abundance and biomass. The more sensitive taxa, such as EPT, were more abundant on larger cobble substrates with moderate velocity, conditions that are typical of erosional type habitats. This finding is corroborated by other studies that suggest riffle habitats have a more diverse invertebrate community (Marchetti *et al.* 2011). In these aspects, LCR is similar to other large, moderately productive river systems, where there is generally a decrease in productivity with velocity, and an increase on larger, more stable substrate. Since velocity is directly linked to flow, it is highly probable that some effects of flow regulation must affect the benthic community.

Sampling in 2014 included the second winter sampling event and the findings were very similar to the 2013 winter sampling event. In comparison with summer and fall, the winter abundance was less, but the winter biomass was the largest of the seasons. Although the benthic invertebrate data typically varied between years, the 2014 data fell within the range of previous sampling. Percent EPT was a diversity measure that was consistently lower in winter compared to other sampling seasons. In contrast, percent *Diptera* and percent *Chironomidae* tended to be higher during winter. Three taxa including *Simuliidae* (black fly), *Simulium* (black fly) and *Orthocladius* (non-biting midge; *Chironomidae*) comprised approximately 85% of the samples. *Chironomidae* and EPT are indicator groups used to measure community balance. Typically an even distribution of Chironomidae, Ephemeroptera, Plecoptera and Trichoptera indicates good biotic conditions. Populations with enhanced numbers of Chironomidae in relation to EPT indicate environmental stress (Shelby 2006). Trichoptera was the dominant group in both the summer and fall comprising 56.3 and 73.4 percent of the relative biomass. In contrast, Trichoptera during the winter comprised less than 1%. Diptera (86.2%) and Ephemeroptera (9.1%) maintained the greatest relative biomass in the winter.

The decline in EPT and enhanced Chironomidae during the winter may coincide with the elevated presence of Didymo or may be due to a normal annual shift. Shelby (2006) reported a reduction in the number of different invertebrate taxa when Didymo was present. Interestingly, the maximum species richness of benthic invertebrates during winter was 29 and 28 in 2013 and 2014, respectively. This was despite the lower prevalence of Didymo in 2014.

With two years of winter sampling, the benthic invertebrate community during the winter appears to be robust and diverse. Prior to winter sampling, we hypothesized that the benthic community would be less abundant compared to the summer and fall seasons, due to environmental variables such as reduced light and low water temperatures (Marchetti *et al.* 2011). Based on the data collected to date, this did not appear to be true. However, the winter samplers were left in the river for 10 and 12 weeks compared to between 6 to 11 weeks for previous 2008 – 2012 summer and fall sessions. The longer deployments may have increased abundance and biomass numbers, but without more data on winter benthic accrual, it is difficult to speculate further about the effects of additional deployment time.

4.5.2 Winter MWF Flows

This is the first year of the study that statistical modelling was used to directly test the effect of flow period on a suite of benthic invertebrate response variables. This initial modelling exercise identified operations during the MWF flow period as potentially an important predictor of percent EPT, Simpson's Index, and species richness. These response variables decreased with an increased water elevation difference between MWF spawning (Jan 1-21) and MWF incubation (Jan 21 – Mar 31). This means that large drops in elevation result in subsequently less EPT taxa, diversity, and species richness. This result corroborates our previous assertions that implementation of the MWF flow regime does have an effect on the benthic invertebrate communities in LCR. Other factors or total production, such as abundance and biomass did not appear to be related to operations as significantly. Continued sampling during the MWF flow period will help identify specific trends and further our understanding of the potential effects of altered flows. But, based on the data collected thus far, we continue to tentatively reject the null hypothesis that the continued implementation of MWF flows does not affect the biomass, abundance and composition of benthic invertebrates in LCR.

4.5.3 Spring/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 appeared to have increased biomass when compared to deeper areas in the river. From this, it appears that the reduction in decline of channel elevation during the RBT flow period, at minimum, stabilizes flows and prevents desiccation events that negatively impact invertebrate and RBT redd survival. However, the larger effect of increasing freshet flows overshadows any possible effects of the RBT flow operating regime. Statistical modeling data during the summer sampling period suggests that freshet is a predominant feature. The models did provide indication that flows, particularly highly variable flows with increasing cumulative drops in elevation, are also likely affecting the benthic invertebrate community. We still hypothesize that the reduction in substrate dewatering during the RBT flow period due to stabilized flows has an effect on the benthic community. We therefore preliminarily reject the null 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

During the FFF period, flows are actually quite stable. These stable flows during the fall resulted in benthic community establishment that was similar to that of a more natural system. In the observed scenario, 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 MCR (Schleppe *et al.* 2013), with the most significant influences occurring in areas that were frequently dewatered. However, since LCR sampling only occurred in permanently submerged areas, estimates of the effect of periodic dewatering on the benthic invertebrate communities in LCR are speculative. Daily variability was not likely great enough to have a large overall effect on the benthic community during the FFF period, but we still speculate that subtle daily changes in velocity likely affect the invertebrate community to some extent and preliminarily reject the hypothesis because flow operations affect physical parameters that affect invertebrates.

4.6 Food for Fish

Many components of the periphyton are good food for benthic invertebrates and subsequently key diet items for fish. The diversity of erosional, depositional and mixed sites in LCR provides a range of feeding opportunities for benthic invertebrates. Additionally, drifting algae from reservoir releases and from periphyton sloughing also provides food. Drift forage quality was lower than the periphyton quality, but was still important to filter feeders such as the web-spinning caddisflies. Of the true periphyton taxa not donated from the drift, the forage quality ranged from good to poor. Most periphyton diatoms provide good forage. Large filamentous green species may not be directly edible, but they create microhabitats that can harbour key food organisms. Unlike Didymo filaments, moderate growths of green filamentous algae are beneficial to LCR productivity (Biggs 2000, Bunn and Arthington 2002).

One of the major features of the annual periphyton cycle in LCR is the proliferation of Didymo during stable winter flows, during riverine conditions we have not yet identified. The muco-polysaccharide Didymo filaments can be problematic due to their low value to grazing invertebrates and resistance to decomposition (Shelby 2006). Many authors corroborate our field observations that Didymo alters periphyton growth beneath the filament mat, and negatively affects benthic invertebrate diversity and density (Mattson 2009; Saffran and Anderson 2009; Shelby 2006). If our hypothesis is correct and Didymo takes advantage of stable winter low flows, then these low flows may be deleterious to benthic invertebrate production, particularly in the mid-depth cobble substrates that become coated with its filament mats. The causes of the lower Didymo production in winter 2014 compared to 2013 are not fully understood, but may relate to the timing and ramping of decreased flows and/or water temperature.

Regression modelling indicated that velocity and substrate score were important determinants of the benthic invertebrate community that is considered high quality forage for fish. Although there was some variation by flow period, high quality forage was positively associated with both velocity and substrate size. Similarly, velocity is also an important determinant of the periphyton community. However, it is important to note that our samplers were biased towards the more productive margins of the river, meaning that sites with very high velocity were not sampled. In extremely high velocity, turbulent sites, we expect to see a reduction in available forage for fish. We continue to tentatively reject all four null hypotheses because operational changes during the MWF, RBT and FF flow periods have a downstream effect on velocity and ultimately the availability of food for fish by adjusting the margins of the highly productive areas of the channels.

5.0 RECOMMENDATIONS

1. Previously, we recommended that the hypotheses associated with Physical Habitat MQ 3 be eliminated or altered, as the current sampling program does not directly test the question and the likelihood of finding an effect of BC Hydro flow management on water quality is low. This was the first year that we attempted to directly test the effect of flow management using statistical analyses on LCR water quality. The analysis was improved over previous years because we were able to incorporate upstream factors such as ALR nutrient addition and ALR water quality, noting that we used many assumptions to include this data. Although the relationships were not particularly strong, it is still highly probable that upstream factors have a direct influence and our modelling needs to develop more appropriate explanatory variables to better elucidate these effects. For example, next year we plan to model water quality analyses at AR8 versus WQSI1, as well as T-N and T-P against magnitude of flow. Also, in the future, attempts should be made to obtain additional nutrient data from more immediate sources such as Celgar and City of Castlegar municipal effluents, as they may also be informative predictors of LCR electrochemistry and biologically active nutrients.

2. In 2013 and 2014, the spring water quality sampling event occurred at the beginning of June, approximately one month before peak freshet. An additional spring water quality sampling date should be added to late June to get closer to peak freshet without the risk of sampling after peak in the falling leg of freshet. This would allow a better understanding of freshet impacts on turbidity, electrochemistry and nutrient concentrations while not overly affecting existing sample size. This recommendation could be easily implemented in 2015 as productively sampling will not overlap with water quality sampling.

3. Ideally winter accrual sampling should be continued at least two more years to further hone in on the peak accruals. Accrual sampling should also be continued in the fall to determine exactly when after 8 weeks peak biomass occurs. Finally, although summer accrual has been done for three years, it might be advisable to repeat this analysis to ensure that the trends previously identified are accurate. While this is less important, we acknowledge that methods between 2008 – 2010 were slightly different than those used between 2011 to 2014.

4. Ideally, productivity sampling should be increased to occur on an annual basis. The high inter-annual variation that has been observed and the identified subtle effects of flow management will require a large sample size to accurately estimate potential effect size. By increasing sampling frequency, the potential effect size may be better estimated.

5. A coordination effort between projects might help develop better, 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 for the FFF). 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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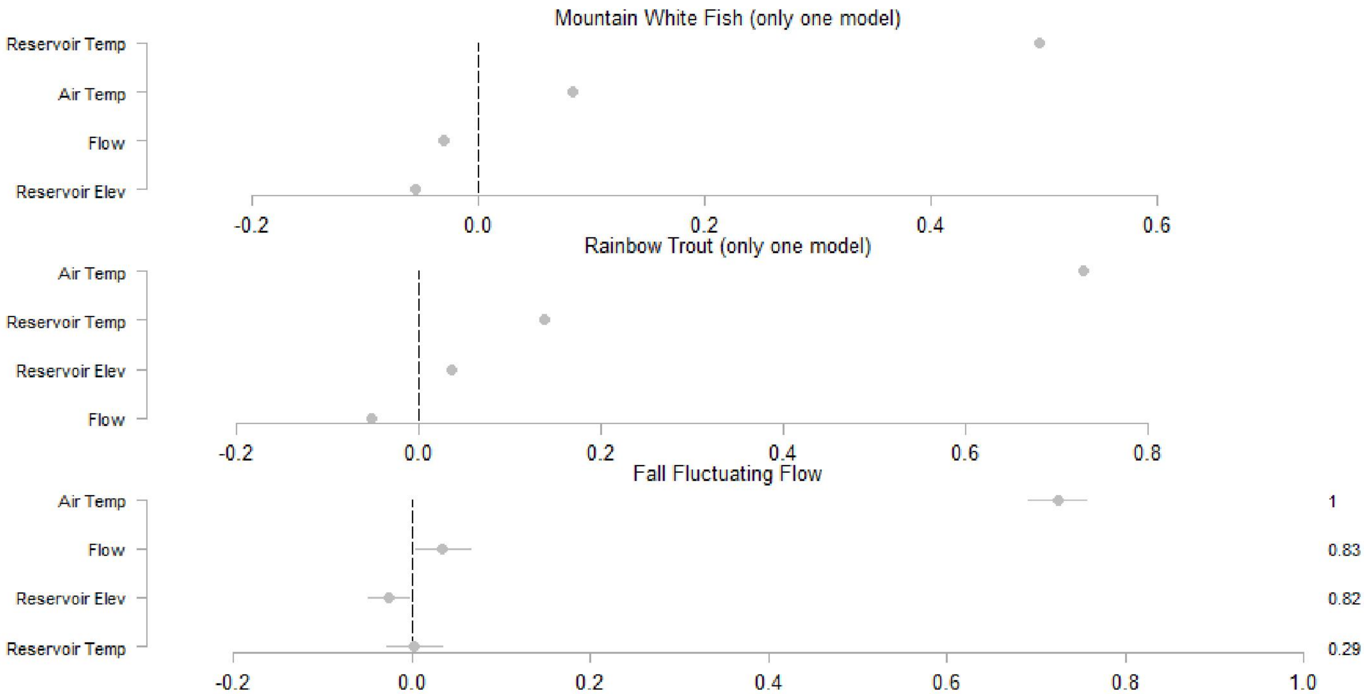
Appendix A – Supplemental Results

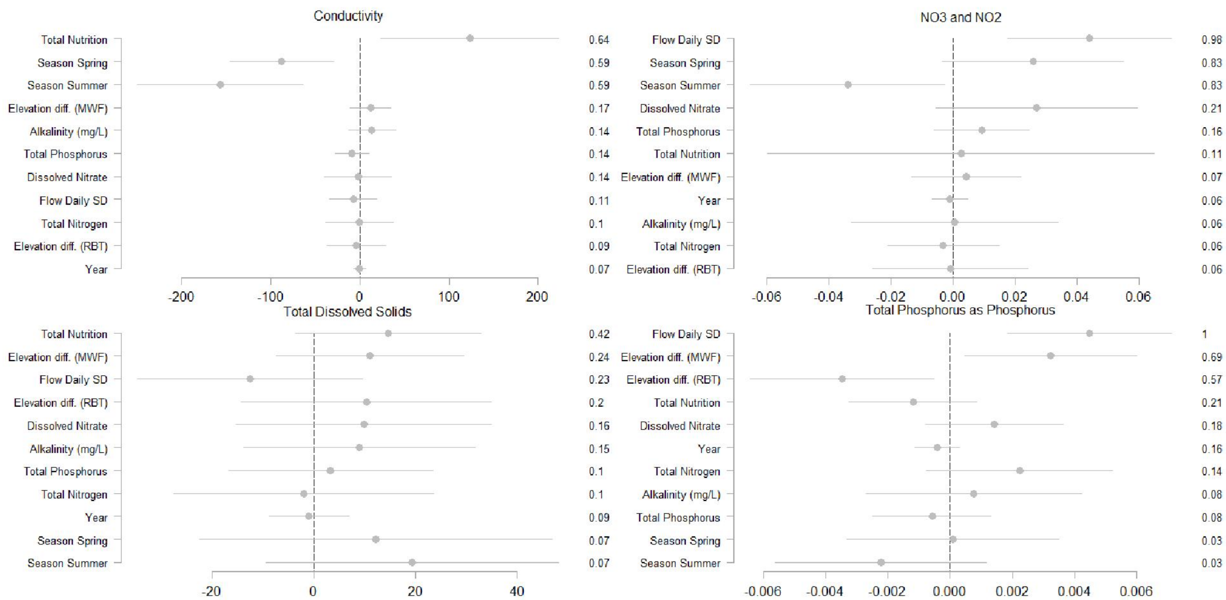
A-1: Mean Daily Flows in 2014 by Designated Flow Period (m3/s)

Year	Statistic	HLK/ALGS	Brilliant	Birchbank
Mountain Whitefish Flows (Jan 1 - Mar 31)				
	N (days)	90	90	90
	Minimum	569.8	451.6	1056.3
	Maximum	1606.1	667.2	2153.9
2014	Median	990.7	490.2	1577.1
	Arithmetic Mean	1037.0	508.8	1576.9
	Standard Deviation	260.6	64.6	273.8
	Coefficient of Variation	0.25	0.13	0.17
Rainbow Trout Flows (Apr 1 to Jun 30)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	91	91	91
	Minimum	500.6	650.0	1223.5
	Maximum	1262.6	2535.9	3632.3
2014	Median	623.8	1704.2	2465.0
	Arithmetic Mean	656.2	1685.3	2409.5
	Standard Deviation	148.9	549.8	691.9
	Coefficient of Variation	0.23	0.33	0.29
Fall Fluctuating Flows (Sep 1 to Oct 31)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	61	61	61
	Minimum	568.5	401.0	1022.4
	Maximum	1356.6	664.9	2048.3
2014	Median	1023.6	404.1	1480.8
	Arithmetic Mean	982.0	421.2	1458.3
	Standard Deviation	265.5	59.0	289.3
	Coefficient of Variation	0.27	0.14	0.20



A-2: 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).



A-3: Scaled and Centered Parameter Estimates (circles) with 95% Unconditional Confidence Intervals (lines) from Averaged Predictive Linear Mixed-effects Models of Electrochemistry and Nutrients in LCR.

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 had side of each figure).

A-4: 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	0.47-0.50	8	0.13-0.18	4	0.60-0.62
Biovolume	6	0.51-0.53	4	0.34-0.35	4	0.58-0.62
Chlorophyll-a	10	0.13-0.16	4	0.44	4	0.41-0.44
Species Richness	8	0.71	4	0.41	5	0.35-0.37
Simpson's Index	10	0.21-0.24	6	0.18-0.19	2	0.38
Percent Community from Reservoir	14	0.00-0.03	13	0.36-0.39	7	0.09-0.13

A-5: 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 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	8	0.55-0.57	7	0.12-0.16	3	0.17-0.26
Biomass	4	0.43-0.44	8	0.005-0.06	8	0.01-0.05
Hilsenhoff Biotic Index	8	0.62-0.64	3	0.41-0.44	10	0.38-0.44
Species Richness	8	0.33-0.38	4	0.27-0.30	5	0.33-0.41
Simpson's Index	2	0.50	6	0.17-0.20	2	0.51-0.52
Percent Chironomidae	8	0.20-0.22	3	0.14-0.17	7	0.01-0.08
Percent EPT	4	0.26-0.27	5	0.07-0.08	5	0.29-0.34
Percent Quality Fish Food	6	0.20-0.21	8	0.07-0.10	5	0.34-0.37