Columbia River Project Water Use Plan

Lower Columbia River Fish

Implementation Year 2012

Reference: CLBMON#44

Lower Columbia River Physical Habitat and Ecological Productivity Monitoring (Year 5)

Study Period: 2012

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Lower Columbia River Physical Habitat and Ecological Productivity Monitoring (Year 5)

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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
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
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) eg. 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 between the outflow of the Hugh L. Keenleyside Dam and the Birchbank gauging station. This report is focused on 2012 data, but also includes summary data collected from 2008 to 2011. This study aims 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 preliminary results.

LCR flows came from the Hugh L. Keenleyside Dam (52.2%) and from the Kootenay River (48.5%) in 2012. Freshet flows surpassed those previously recorded and the peak flow occurred later than typical (6,043.1 m^3/s on July 21th 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 flows. Similarly, cumulative elevation drops that occurred during pre-RBT flows were significantly higher than those determined during post and continuous flow periods. Water temperatures varied seasonally, ranging from approximately 4 to 18°C in 2012. Regression modeling 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 three occasions in 2012 and indicated good water quality in both the Kootenay and LCR. We have yet to test whether MWF, RBT or FFF alter the availability of biological active nutrients and/or the electrochemistry of the river, however, we expect that the influence of these managed flow periods on water quality will be subtle compared to the overwhelming effects of freshet, anthropogenic nutrient donation or alteration, and photosynthesis. 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 summer, fall and winter using artificial substrate samplers that were deployed on the river bottom for between 10 to12 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 water temperature, substrate type and size, erosional or depositional channel characteristics, depth and light at the deployment sites. Seasonal patterns and annual variation had the greatest impacts on periphyton and benthic invertebrate community structure. However, the specific effects of the MWF and RBT flow periods were difficult to separate from larger scale effects such as the seasonal freshet pattern. The extremely high flows documented in the summer of 2012 temporarily reduced the ecological productivity of river.

Preliminary statistical modeling showed that the availability of food for fish was also affected by season. High peak freshet flows during the RBT flow period apparently reduced food availability for fish in erosional areas, but during the FFF, the availability of food for fish was greatest in erosional cobble banks with higher velocities and included more of the sensitive Ephemeroptera, Plecoptera and Trichoptera taxa. We hypothesize

that fish food availability declined in winter 2013 due to the seasonal establishment of Didymo mats and less favorable habitat conditions. EPT taxa were not as prevalent in substrates coated with Didymo filaments, while their numbers increased in mid-channel areas where higher water velocities reduced Didymo densities. The observation that stable winter MWF flows may encouraged excessive growth of Didymo, reducing densities of EPT taxa (high quality fish food) will be further explored during future sampling events.

Table 1-1: CLBMON-44 Status of Objectives, Management Questions and Hypotheses After Year 5

Management Questions	Management Hypotheses	Year 5 (2012) Preliminary 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 such as air temperature, reservoir temperature and reservoir elevation. The strength of the relationship between flow and LCR water temperature was similar for each of the flow periods and for the winter and summer seasons. These findings are consistent with that reported by Scofield et al. (2011) for years 1-3 of the study. Given the nominal influence of flow on LCR water temperature, the null hypothesis is preliminarily accepted.
Physical Habitat Monitoring Q.2. How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect the seasonal and inter-annual range and variability in river level fluctuation in LCR?	Ho2phy: Continued implementation of MWF and RBT flows does not affect seasonal water levels in LCR. Ho2Aphy: Continued implementation of MWF flows does not reduce the river level difference between the maximum peak spawning flow (1 Jan to 21 Jan) and the minimum incubation flow (21 Jan to 31 Mar). Ho2Bphy: Continued implementation of RBT flows does not maintain constant water level elevations at Norns Creek fan between 1 Apr and 30 Jun.	Regression modeling suggests that river flow is an important determinant of water levels. River elevation data at five sites on LCR were regressed with flow data from HLK, BRD, and BBK. Historic flows and the linear relationships were then used to predict elevations during pre, post and continuous MWF and RBT flow periods. The elevation differences during the flow periods were analyzed using permutation ANOVA. At all locations, the river level difference between MWF maximum peak spawning and minimum incubation was greater during pre-MWF flows than with post and continuous flows. Similarly, river elevation data from WQIS2 and 3 were regressed with flow data during the RBT flow period. The best fit regression model was used to predict historic elevations For both WQIS, the cumulative elevation drops that occurred during pre-RBT flows (1984-1991) were significantly higher than those determined during post (1992-2007) and continuous (2008-2012) flow periods (ANOVA: WQIS2, d.f. 2, 26, $p<0.001$; WQIS3, d.f. 2, 26, $p=.0028$). Based on the data collected to date, we preliminarily reject all three null hypotheses.
Physical Habitat Monitoring Q.3. How does continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall affect electrochemistry and biologically active nutrients in LCR?	Ho3phy: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the water quality of LCR. Ho3Aphy: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, does not alter the electrochemistry of LCR.	Water quality parameters that address electrochemistry include conductivity, TDS, hardness, alkalinity, dissolved metals ions and pH. Based on data collected in all years, LCR had good water quality. Parameters rarely exceeded water quality guidelines or objectives. The 2012 electrochemistry data corroborated earlier years of this study, but also revealed relationships between dissolved constituents and dilution during peak flows. Biologically active nutrients include; nitrate, ammonia, total P and ortho phosphate (SRP). The frequency of non-detectable nutrient results over the years of study weakened the averages. Preliminary data

	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.	suggested that Reach 2 of LCR should be classified as oligotrophic and the Kootenay River as mesotrophic. Given the data collected thus far, we have been unable to directly test whether MWF, RBT or FFF alter the availability of biological active nutrients and/or the electrochemistry of LCR. We expect the influence of fish flows on water quality to be subtle compared to the overwhelming effects of freshet, anthropogenic nutrient donation and photosynthesis. We anticipate that fish flows could 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 LCR. Future water quality sampling will be undertaken during the winter months to better address the MWF flow period hypothesis, and all water quality hypotheses will be addressed in future years when more data is available.
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. 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.	The high seasonal and annual variation observed in the benthic invertebrate data makes it difficult to attribute a causal effect to the MWF, RBT, or FFF periods. It is hypothesized that stable flows during the MWF period may aid in the establishment of Didymo and have subsequent effects on the benthic community, but additional years of data are needed to confirm this association. During the RBT flow period, the effects of freshet were greater than potential effects of the flow management regime intended to reduce cumulative drops in river elevation. Despite this, we still hypothesize that the reduction in substrate dewatering during the RBT flow period has acted to stabilize flows and the invertebrate community. During the FFF period, the effects of dewatering likely causes similar biomass loss to those documented in MCR (Schleppe *et al.* 2013), with the most significant influences occurring in areas that are frequently dewatered. However, since LCR sampling only occurred in permanently submerged areas, changes to the peripheral community are difficult to ascertain. At this time, we preliminarily reject all four null hypotheses because at minimum, flow management has resulted in changes to the LCR benthic invertebrate community. In future years, we will attempt to elucidate the specific effects of the MWF, RBT, and FFF periods on the benthic community.
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	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. Ho2Aeco: Continued implementation of MWF does not increase total biomass accrual of periphyton in LCR.	Based on the periphyton data collected thus far, it appears that the management of flows have the potential to alter the periphyton community, with the specific effect dependent on the flow period in question.

abundance, diversity, and biomass of benthic invertebrates?	Ho2Beco: Continued implementation of RBT flows does not increase total biomass accrual of periphyton in LCR.	In 2012/13 the low stable MWF flows during the winter enabled extensive Didymo growth which lowered periphyton forage quality, but contributed to very high productivity metrics. We therefore preliminarily reject Ho2Aeco, that MWF flows does not increase total biomass accrual of periphyton. In 2014, periphyton sampling during the MWF flow period will include the collection of weekly Chl-a accrual data, which will provide additional information to more thoroughly address this hypothesis. The combination of large spring freshet and RBT flows lowered LCR periphyton productivity in 2012. Lower summer periphyton production metrics compared to the fall are consistent with results reported by Scofield et al. (2011) for years 1-3 of this study. Because we cannot separate the effect of spring freshet from RBT flows, we tentatively accept Ho2Beco, that RBT flows does not increase total biomass accrual of periphyton. Mean daily flows during the FFF period were generally quite stable. These moderate flows typically allowed more periphyton growth compared to the spring/summer. We preliminarily accept Ho2Ceco, that FFF do not increase total biomass accrual of periphyton in LCR, since the fall biomass data was typical of large rivers.
	Ho2Ceco: Continued fluctuations of flow during the fall do not increase total biomass accrual of periphyton in LCR.	
Ecological Productivity Monitoring Q.3. Are organisms that are used as food by juvenile and adult MWF and RBT in LCR supported by benthic production in LCR?	Ho3eco: Continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not increase the availability of fish food, organisms in LCR	A fish food index (FFI) was used to evaluate the effects of flow management on food for fishes in LCR. The final FFI score for each site represents the abundance of benthic taxon as fish food, the size or biomass availability of benthic taxon as fish food, and the availability of more preferred types of benthic foods. Modeling data suggest that high peak flows during summer periods, as was observed in 2012, had an overall negative effect on food for fish. However, during more stable periods, food availability for fish was positively associated with velocity. This suggests that during the RBT flow period, high peak freshet flows reduce food availability for fish, making detection of specific effects associated with the RBT flow regime more difficult. During the fall fluctuating flow period, the availability of food for fish was greatest in areas of higher velocity. Areas of higher velocity were more typical of erosional, cobble banks , which tended to have greater predominance of the more sensitive EPT taxa. Although not specifically modelled, we hypothesize that fish food availability will decrease in the winter due to the establishment of Didymo and less favourable habitat conditions. We preliminarily reject all four null hypotheses because at minimum, flows appear to affect the availability of food for fish. In future years we will attempt to better understand the specific effects of the MWF, RBT, and FFF operating regimes on food for fish.
	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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APPENDICES

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. Dam operations on large rivers have been shown to have downstream ecological implications (Gregory et al. 1991, Allan and Flecker 1993, Blinn et al. 1995), and LCR is no exception (Hildebrand 2009, Watts 2009, Baxter and Thorley 2010). Over the past decade, British Columbia 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.

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 5 (2012) of the study and includes both historic and 2012 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);
2) Rainbow Trout (RBT) protection flows (Apr 1 – Jun 30); and
3) fall fluctuating flow (FFF) (Sep 1 – Oct 31).

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, including water quality index stations (WQIS) and benthic productivity sampling stations. Physical parameters including water quality, water temperature and water level were collected at the 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).

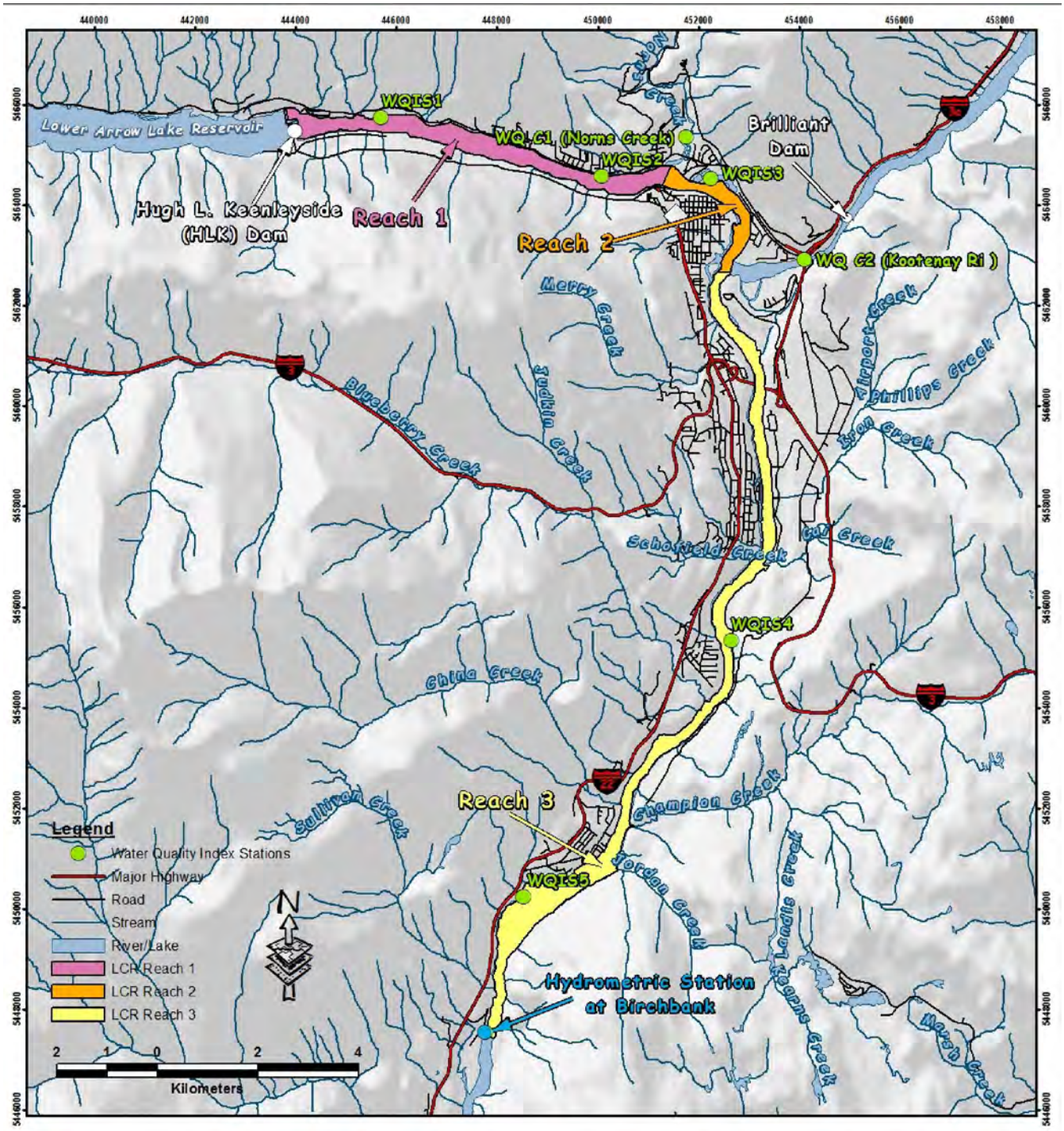

Figure 2-1: Map of Lower Columbia River Study Area and Water Quality Index Station Sampling Locations

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 2-2: Benthic Productivity Sampling Locations in 2012/13.

2.2 Hydrology and Water Level

Water level and temperature data were collected at five water quality index stations (WQIS) 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 (ALGS), 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 2012, 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 water quality index stations (WQIS1 through 5) on LCR and at one station on Kootenay River (WQ C2) (see 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. The barologger 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. This report includes river stage data collected between December 2011 and December 2012.

2.3 Physical and Chemical Characteristics

Chemical and physical water quality parameters were collected at seven different sampling locations during 2012 (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

[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.

(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 LCR WQIS occur in proximity to noteworthy nutrient sources. WQISI occurs in close proximity 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).

During 2012, field trips were conducted on June 1, August 14, and October 25, 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 SpeedTech hand-held 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.

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. 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, B.C. 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 Year 5 differed from Years 1-3, in that all sampling locations were located in Reach 2 and were sampled during summer and fall in 2012 and during winter in 2013. 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 in 2012/13 were 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
D	Deep	>5.5 m
MD	Moderately deep	4 – 5.5 m
M	Mid	2.5 – 4 m
MS	Moderately shallow	1 – 2.5 m
S	Shallow	<1m

2.4.1 Periphyton Natural Substrate Sampling, Drift Sampling and In Situ Experiments

Quantitative natural periphyton samples were collected from sites adjacent to the artificial samplers in August 2012 and March 2013. The species composition of samples were compared to those from the artificial substrate samplers to understand the extent of artificial substrate bias on species composition, if any, and to obtain a more comprehensive algal and invertebrate species list for LCR. The methods used were designed to minimize natural variation and they conform to the USEPA Rapid Bioassessment Protocol for Periphyton (Chpt.6, Barbour et al. 1999). Sample areas had similar water depth, flow patterns, velocity, substrate size, and minimal macrophyte cover or riparian shading, and were water-covered for at least 10 weeks as of the time of sampling.

For erosional (cobble) sample sites, five smooth cobbles exceeding 20 cm were selected and placed on a plastic tray at the river's edge to minimize drying. In brief, a sampling cylinder fitted with a rubber gasket was held firmly on the apex surface of the cobble, then a scalpel, brushes and a squirt bottle of filtered water were used to remove all the periphyton within the sampler diameter using 100 mL from each cobble after rock and funnel were rinsed into beaker (coarser sand and predators were noted but not added to sample).

For depositional (sand/silt) sample sites, samples were collected from wadeable depths (generally 0.5 – 0.75m) that had been water-covered for at least 10 weeks. A large petri dish was inverted onto the surface sediment, a rigid spatula was inserted under the dish to obtain a known volume. The sampled substrate was scraped and rinsed into a 300 mL bottle, filled with filtered river water, shaken vigorously for 60 seconds before a 10 mL subsample was decanted. This process was repeated 5 times for each of three replicate samples per site (15 petri dish sediment samples per site).

On each of the field trips, drift samples and 1L whole plankton samples were collected from the furthest upstream and downstream LCR sites (WQIS1 and WQIS5), and from Kootenay River (WQ C2). Qualitative drift samples were collected by suspending a standard 80-micron mesh plankton net for one minute in the upper water column. The drift samples were immediately cooled on ice (not frozen) to limit predation within the sample. On two dates, 4-liter composite chlorophyll-a (chl-a) samples were collected at the same sites. They were placed in the dark and on ice immediately upon collection and were also delivered to Caro Labs within 24 hours.

Periphyton grab samples (non-quantitative) were collected from selected sample sites, with an emphasis on *Didymosphenia geminata* (Didymo). These composite non-quantitative samples were collected from a variety of substrates and transferred to zip-lock bags, and then stored on ice or refrigerated prior to analysis. Grab samples of aquatic macrophytes were also opportunistically collected when encountered. They were also chilled in large Ziploc bags prior to pressing and mounting for a herbarium collection.

During the summer and fall sampling sessions, an additional *in situ* experiment with alternate substrates was conducted in LCR. Honed stone tiles were attached with silicone adhesive to a Plexiglas strip that was then randomly mounted to a number of the sampling frames. At the end of the sampling session, tiles were pried loose and slipped into a marked plastic bag, and transported in a cooler on ice. In the lab, the entire stone tile surface was scrubbed with a dental cleaning tool. Enumeration followed the same protocols as other periphyton samples, and periphyton densities were calculated based on the size of the tile sampled.

A combination of natural and artificial substrate (Styrofoam and stone tile) sampling was used to comprehensively address the first productivity management question regarding the composition, abundance, and biomass of epilithic algae and benthic invertebrates.

All 2012 periphyton taxonomic identification was completed by Heather Larratt, R.P.Bio. Algae samples were settled in counting chambers over 24 hours. Cells were counted along mid-section transects examined at 400X- 800X magnification with a phase contrast inverted microscope. Counts continued until the relative abundance of taxa stabilized or 300 - 500 cells had been counted. Approximate cell biovolumes were calculated for each sampling campaign using a micrometer, and compared to reported sizes. All parts of the microflora were evaluated, including detritus, vascular debris, bacteria, fungi, yeasts etc. and their micro-grazers (protozoa). Microscope photographs of typical assemblages were shot from each sample and archived for BC Hydro.

Finally, sampling for invasive dreissenid mussel larvae (veligers) was conducted. This involved combining multiple one minute tows using a standard 67micron plankton net near boat launches.

2.4.2 Periphyton and Invertebrate Community Sampling using Artificial Samplers

2.4.2.1 Artificial Sampler Design and Deployment

The artificial sampler design was substantially modified from the first three years of the study. The latest design is similar to that employed in the middle Columbia River, below the Revelstoke Dam for CLBMON-15b. Reasons for the altered design included recommendations from Scofield et al. (2011) who identified a problem with deployment ropes rubbing against the artificial substrate and possibly disrupting periphyton growth. Immediately prior to the summer 2012 deployment session, the artificial sampler design was altered again to accommodate SARA permit #245 which was issued on May 29, 2012 under Section 73 of the Species at Risk Act. This permit allowed for shorthead sculpin (*Cottus confusus*) to be incidentally collected, then released unharmed at the site of capture. A condition of the permit was to avoid disturbing potential residences of shorthead sculpin by not retrieving the baskets until after August 15th. This altered retrieval time made data comparisons with previous years difficult, as the summer sampling session was 11 weeks long, rather than 8. The sampling design (Figure 2-3) that

was utilized in the summer and fall, allowed for the retrieval and redeployment of the periphyton sampler without disturbing the rock basket. The rock basket was positioned adjacent to the steel anchor at a distance more than 20 m from the periphyton sampler. The downfall of this design was that more rope was needed and the apparatus was more difficult to deploy. With this design, we planned to retrieve the periphyton samplers after 8 weeks, take the necessary periphyton punches for analysis and comparison with previous years, and then redeployment them until August 15, when both the periphyton and benthic invertebrate samplers would be sampled and removed from the river. In the end, this plan was not implemented due to extremely high flows in July and early August of 2012. After discussions with BC Hydro Project Managers, it was decided that the 8-week retrieval would not be undertaken due to safety concerns surrounding the high flows.

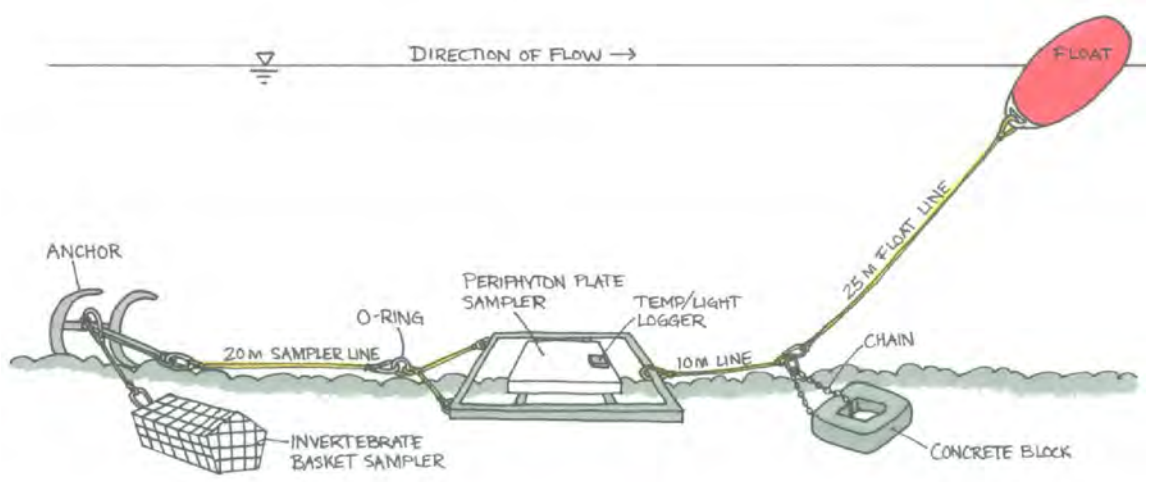

Figure 2-3: Diagram of the Periphyton and Macroinvertebrate Sampling Apparatus Deployed in the Summer and Fall to Accommodate Short-head Sculpin

For the fall incubation period, substrates were deployed in the middle of August and retrieved during the third week in October, for a 10-week incubation period. Previously, the fall incubation period was 8 weeks, but Scofield et al. (2011) recommended that the duration of the fall sampling period be extended to capture peak periphyton biomass.

For the winter sampling session, the original apparatus design was used, because it was easier to deploy in the harsher winter conditions (Figure 2-4). For the winter incubation period, substrates were deployed from January 9th through the end of March, for a 12-week incubation. This sampling period was designed to coincide with MWF flows, and given the cooler temperatures, it was expected that primary and secondary production would occur at a slower rate when compared to summer or fall sampling (Marchetti et al. 2011).

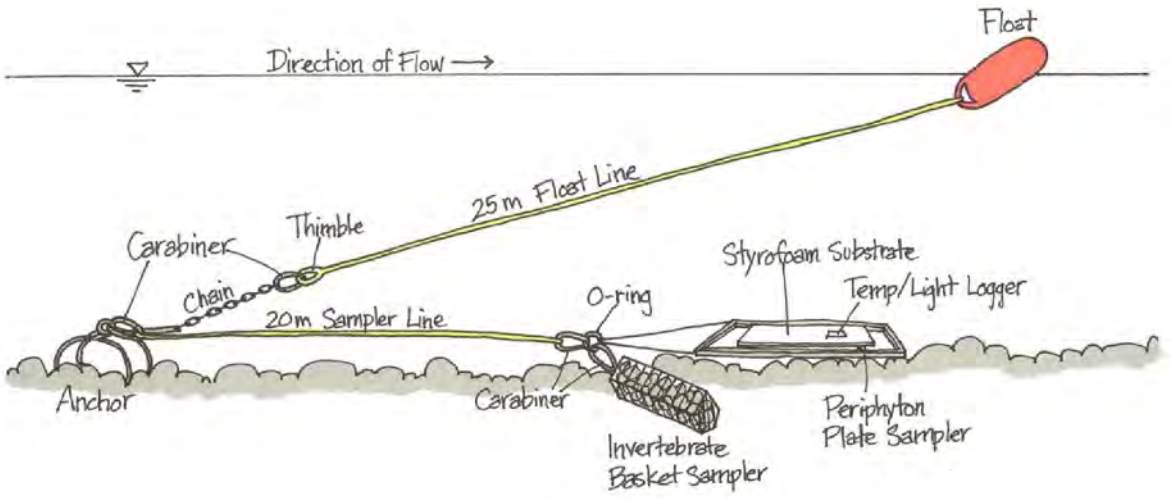

Figure 2-4: Diagram of the Periphyton and Macroinvertebrate Sampling Apparatus Deployed in the Winter of 2013 (Jan – Mar)

The high flows and turbid conditions at the time of the fall deployment in mid-August necessitated some creative thinking regarding sampler deployment at the necessary depths when the bottom of the river could not be seen. To ensure the samplers were deployed right side up, we utilized a chandelier method of deployment (Figure 2-5). 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 2-5: The Chandelier Deployment Method

In summary, the duration of the 2012/13 summer, fall and winter sampling sessions were 11, 10 and 12 weeks, respectively (Table 2-3 for deployment dates, sampling numbers and recovery rates). This sampling duration differed from the 8 and 6 week deployments in previous years. There were seven productivity sampling sites within reach 2 (see Figure 2-2 for site locations), and five artificial samplers were deployed at each site. The one exception was no artificial samplers were deployed at Site 2 in the fall due to the number of lost and entangled samplers that were trapped in the river from the summer deployment. Site 2 was particularly problematic with regards to high velocities; three entangled samplers remained trapped at Site 2, therefore it was decided to avoid this site for fall deployment.

At the time of deployment, the elevation and location of each artificial sampler was recorded using a Trimble R8 RTK survey system, using Survey Controller software for data collection to accurately obtain the geodetic elevation of each sampler.

Table 2-3: Artificial Sampler Deployment and Rates in 2012/13 Recovery

Season	Reach	Site	Periphyton Samplers		Invertebrate Basket Samplers	
			# Deployed	# Retrieved (% Recovery)	# Deployed	# Retrieved (% Recovery)
Summer (Jun 1 - Aug 15) 11 weeks	2	Site 1 (S1)	5	5 (100)	5	5 (100)
		Site 2 (S2)	5	1 (20)	5	1 (20)
		Site 3 (S3)	5	2 (40)	5	2 (40)
		Site 4 (S4)	5	4 (80)	5	4 (80)
		Site 5 (S5)	5	4 (80)	5	4 (80)
		Site 6 (S6)	5	5 (100)	5	5 (100)
		Site 7 (S7)	5	5 (100)	5	5 (100)
Summer Totals			**35**	**26 (74)**	**35**	**26 (74)**
Fall (Aug 17 - Oct 24) 10 weeks	2	Site 1 (S1)	5	3 (60)	5	4 (80)
		Site 2 (S2)	0	-	0	-
		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	4 (80)
		Site 6 (S6)	5	5 (100)	5	5 (100)
		Site 7 (S7)	5	5 (100)	5	5 (100)
Fall Totals			**30**	**27 (90)**	**30**	**28 (93)**
Winter (Jan 9 - Mar 30) 12 weeks	2	Site 1 (S1)	5	5	5	5
		Site 2 (S2)	5	5	5	5
		Site 3 (S3)	5	5	5	5
		Site 4 (S4)	5	5	5	5
		Site 5 (S5)	5	5	5	5
		Site 6 (S6)	5	5	5	5
		Site 7 (S7)	5	5	5	5
Winter Totals			**35**	**35 (100)**	**35**	**35 (100)**
2012/13 Totals			**100**	**88 (88)**	**100**	**89 (89)**

2.4.2.2 Artificial Sampler Retrieval

At the time of retrieval, a random number generator was used to take four Styrofoam punches from each sampler to assess the following metrics: 1) Chlorophyll-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.

In addition to the three seasonal incubation periods, a few periphyton plates were left in the river following both the fall and winter incubation periods. The purpose of these plates was to generate long-term periphyton chl-a and taxonomic data to further understand if the incubation periods were long enough to reach peak biomass.

Artificial substrate sampling differed from previous years in that accrual data was not collected on a weekly basis. Given the strength of the accrual data from 2008 – 2010, accrual resources were spent on initiating a winter sampling program. This first year of winter sampling was a trial year with no accrual data collection. However, accrual data will be collected during the winter incubation period in 2014.

2.4.2.3 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 2012 and previous years of the study (2008 – 2010) 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

2.4.2.4 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 were conducted in R (R Development Core Team 2012).

2.5.1 Data Manipulation

Prior to carrying out statistical analyses on data across multiple years, data from the first three years of the study (Scofield et al. 2011) were combined with those collected in 2011 and 2012. In some cases the combining of data required updates to naming conventions, changing of reporting units, etc. Quality assurance and quality control measures were incorporated throughout the data manipulation process to minimize error.

Periphyton production data collected from 2008 to 2010 contained many repeated sampling events at several artificial sampler locations. As the specific sampling method could not be determined, and to avoid pseudo-replication, the mean of the two data points for any given sampler was determined. Only the mean data was used in subsequent analyses.

Flow data from BBK gauging station was several hundred cubic meters/second less than the combined flows of HLK and BRD dams in December 2009. This potential error has been reported to Poisson Consulting for verification. To alleviate the error in data analysis, the sum of HLK and BRD dam flows was used in the analysis to represent BBK rather than the original data.

2.5.2 Water Levels

The mean 2012 water level elevations recorded at WQIS1-5 in LCR were compared to the combined water elevation (± SD) during all years to visualize the effects of high flows observed in 2012. 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.

2.5.2.1 Mountain Whitefish (MWF) Flow Period

The effects of the MWF flow period were investigated by analyzing the water elevation difference between the maximum elevation during spawning and minimum elevation observed during incubation at each WQIS. To determine elevation differences, a linear regression model of flow and water elevation was conducted at each WQIS, using the flows from HLK, BRD, and BBK and their associated quadratic terms as explanatory variables. Akaike information criterion corrected for small sample sizes (AICc) model selection was used to determine the best fit model, although plausible models (those with an AICc <2) were considered. The regressions at each site were 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 (2008-2012). 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 typical ANOVA or Student's t tests because the data met the assumptions of the test and this method was preferred to non-parametric methods. Finally, the data were visually compared to actual elevations measured between 2008 - 2012 to investigate how predicted elevations compared to field collected elevations.

2.5.2.2 Rainbow Trout (RBT) Flow Period

To address the hypothesis that continued implementation of RBT flows does not maintain constant water level elevations at Norns Creek fan between April 1 and June 30, the cumulative drop in water elevation at WQIS2 and 3 were investigated. To calculate the cumulative elevation differences over the RBT flow period, a regression of flow and water elevation was conducted, using the flows from HLK, BRD, and BBK and their associated quadratic terms as explanatory variables. AICc model selection was used to determine the best fit model, although plausible models (those with an AICc <2) were considered. The regressions at each site were used to predict water elevation for time periods between pre-implementation of RBT flows(1984 to1991), implementation of RBT flows (1992 to 2007), and continuation of RBT flows (2008-2012). 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 typical ANOVA or Student's T Tests because the data better met the assumptions of the test and this method was preferred to non-parametric methods. Finally, the data were visually compared to actual elevations measured between 2008-2012 to investigate how predicted elevations compared to field collected elevations.

2.5.3 Water Temperature

We used linear mixed-effects modeling (Zuur et al. 2009) and model selection via Akaike information criterion corrected for small sample sizes (AICc) to evaluate the relative effects of above site reservoir temperature and elevation, flow from dams (HLK and BRD), Castlegar air temperature, and seasonal flow period, and *a priori* hypothesized interactions between flow period and other explanatory variables on LCR water temperatures. We included sampling site and year as random effects in all models to avoid pseudo-replication and to determine relationships across time and sites. We included sampling site and year as random effects in all 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. For this reason, we had to standardize explanatory variables by location. Flow was standardized by using flows from HLK for WQIS1 through WQIS3 sites and BBK flows for WQIS4 and WQIS5 sites. Reservoir temperature and elevation were weighted by flows, where WQIS1 through WQIS3 were divided by HLK flows and WQIS4 and WQIS5 were divided by BBK flows. Temperature data from Kootenay Lake was only available for one to two days in each season. We created a full temperature dataset for this lake to be used in subsequent models by predicting daily water temperature from a Generalized Additive Model for both Kootenay Lake and Arrow Reservoir that included day of year (1-365), season, and location (Kootenay Lake or Arrow Reservoir). This model allowed us to use point data from Kootenay Lake and our full dataset from Arrow Reservoir to better predict seasonal trends in water temperature in subsequent analysis. Data was obtained from numerous different sources and only preliminary exploratory review of data quality was conducted.

We evaluated the relative support for these hypotheses using an all model combinations approach. Model uncertainty was assessed using AICc (Burnham and Anderson 2002; Anderson 2008). The lower the AICc score for a given model, the better the trade-off between complexity and optimal fit for that model. We used the MuMIn package in R (Barton 2012) to compete models based on ΔAICc values and AICc weights (w_i)(Burnham and Anderson 2002). We also calculated pseudo R^2 to understand the variation in slected models. R^2 was derived from regressions of the observed data versus fitted values (see Cox and Snell 1989; Magee 1990; Nagelkerke 1991; Piñeiro et al. 2008 for details).

We used non-standardized continuous explanatory variables to maintain the predictive utility of averaged models. In order to compare among all parameters and interpret the main effects in conjunction with interaction terms, we also conducted the above analyses after standardizing continuous explanatory variables by subtracting global means from each value (centering) and dividing by two times the SD (scaling)(Gelman 2008).

2.5.4 Benthic and Periphyton Community

Non-metric multidimensional scaling (NMDS) using Bray-Curtis dissimilarity was used to explore variation in benthic community composition. Data were transformed using the Bray Curtis transformation. Finally, ANOSIM was used to determine if groups (either Ward's or other meaningful groupings) were significantly different in composition. NMDS was run for the species taxonomic level for benthic invertebrates and the genus taxonomic levels for periphyton. Future community analysis will also use this approach to understand the small and large scale taxonomic community differences by investigating other taxonomic levels of identification.

2.5.5 Benthic and Periphyton Production

Exploratory analysis of production responses to predictors was completed for raw and transformed data. The intent of this step was to reveal any general patterns or trends across transects prior to any statistical analyses.

Four response variables for both periphyton and benthic invertebrates were modeled. Periphyton response variables included: 1) abundance, 2) biovolume, 3) chlorophyll-a, and 4) Simpson's index. Invertebrate production and diversity response variables included: 1) abundance, 2) biomass, 3) Simpson's Index, and 4) Hilsenhoff Biotic Index (HBI). Diversity and production data were transformed using either log10 or square root to adhere to the assumptions of least-squares multiple regression (i.e. normal distribution of residuals and heteroscedasticity of residuals).

Hilsenhoff Biotic Index is typically used as a measure of oxygen concentration in organic loading of rivers, relating water quality conditions to the benthic biota where higher index values indicate low dissolved oxygen conditions and hence poor water quality. The index factors the sensitivity of different taxonomic groups to low oxygen conditions. To some extent, low oxygen conditions originating from poor water quality are similar to extremes associated with different flow operating regimes, as data from the MCR shows (see Schleppe *et al.* 2011). The HBI is calculated as follows:

$$HBI = \sum x_i t_i / n$$

where x_i is the number of individuals within a taxon, t_i 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).

We used linear mixed-effects modeling (Zuur et al. 2008) and AICc model selection similar to that described above to evaluate the relative effects of mean daily light intensity, water temperature, air temperature, Arrow Reservoir water temperature and flows, water velocity, substrate, and relative depth (shallow, moderate shallow, moderate, moderate deep, and deep) on periphyton and benthic invertebrate production response variables in each season with site and year as random effects. We evaluated the relative support for these hypotheses using an all model combinations approach. Model uncertainty was assessed using AICc and multi-model averaging (Burnham and Anderson 2002; Anderson 2008). We used the MuMIn package in R (Barton 2012) to compete 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 0.6 considered to be of high importance in subsequent interpretations. We also calculated pseudo R^2 for high ranking models (derived from regressions of the observed data versus fitted values)(see Cox and Snell 1989; Magee 1990; Nagelkerke 1991; Piñeiro et al. 2008 for details) which give an indication of the proportion of the variance in response variables explained by individual models.

We used non-standardized continuous explanatory variables to maintain the predictive utility of averaged models. to compare among all parameters and interpret the main effects in conjunction with interaction terms, we also conducted the above analyses after

standardizing continuous explanatory variables by subtracting global means from each value (centering) and dividing by two times the SD (scaling)(Gelman 2008).

2.5.6 Fish Food Index

A FFI was calculated using three criteria to assess how the effects of flow and physical conditions on LCR may affect food for fishes. The criteria were: 1) abundance of invertebrate taxa, 2) biomass of invertebrate taxa, and 3) a ranking of preference as fish food for different invertebrate taxa. The index is conceptually similar to the index developed by Sass et al. (2011). The Fish Food Index is calculated as follows:

$FFI = \sum_1^I \ \%A \times \%B \times P$

where:

1 throμgh the I^{th} is the benthic taxon at any given sampling site,
% A is the percent abundance of any given benthic taxon;
B is the biomass of the taxon / total benthic invertebrate biomass at the site; and,
P is a fish food preference.

Biomass was determined by multiplying the average weight per individual for larger benthic groups by benthic abundance. Biomass was calculated in this manner because biomass was only measured by group rather than by individual taxon.

For each benthic taxon encountered, a fish food preference rank was assigned. The fish food preference rank was assigned using stomach contents data from MCR, literature reviews, and professional judgment of foraging behaviors.

The fish food preference rankings were determined for each different fish species / life stage of importance identified for the MCR. Finally, the fish food preference for the analysis was determined by averaging the score of all fish species / life stages. Future calibration of the fish food preference values can be completed as more data becomes available, or the FFI can be used to consider only specific fish species / life stages. MCR fish food preference rankings are being used because a specific preference ranking for LCR had yet to be developed. The use of MCR rankings is not believed to have a substantial effect on the overall outcome of subsequent modeling.

The final FFI score for each site represents the abundance of benthic taxon as fish food, the size or biomass availability of benthic taxon as fish food, and the availability of more preferred types of benthic foods. The FFI score ranges from 0 to variable maximum (depending on the number of species), with higher overall scores indicating a greater presence of benthic invertebrates preferred as fish food. Future iterations of the index will attempt to standardize scores within a specified range to help aid interpretation of results and facilitate comparisons to other rivers or data collected by BC Hydro. The FFI score for each site was subsequently modeled using the same independent variables as other models for periphyton and invertebrates.

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 2012, the mean daily river flows from the Columbia and Kootenay Rivers were 52.2% and 48.5%, respectively, of the total flows at the Birchbank gauging station. This constituted 98% of the total flow, with the remaining 2% originating from smaller tributaries (e.g. Norns Creek) and outfalls.

Figure 3-1 depicts the 2012 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 (1353.6 m^3/s compared to 1188.8 m^3/s), but Brilliant exhibited a higher peak, with a maximum flow of 3424.4 m^3/s recorded on July 2nd (Table 3-1).

The highest flow recorded at the Birchbank gauging station in 2012 was 6,043.1 m^3/s on July 21th. This is compared to 2011 when the peak was 4,155.4 m^3/s on July 9th (Olson-Russello et al. 2012).

Table 3-1: Mean Daily River Flows (m^3/s) at HLK Dam, Brilliant Dam and the Birchbank Gauging Station in 2012

Location	N (days)	Statistic	2012
HLK	366	Mean	1353.6
		Min	568.2
		Max	3258.0
		SD	646.4
Brilliant	366	Mean	1188.8
		Min	131.2
		Max	3424.4
		SD	886.7
Birchbank	365	Mean	2593.8
		Min	1078.7
		Max	6043.1
		SD	1211.4

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, (Table 3-2). During the MWF flow period (Jan 1 – Mar 31), flows at HLK Dam, Brilliant Dam and the Birchbank gauging station were less variable than those at other times of the year (Figure 3-1). Mean daily flows at the HLK Dam showed a modest downward trend over

the MWF flow period, which ranged from approximately 1850 to 950 m^3/s, while flows at Brilliant Dam increased slightly with a change in flow of less than 550 m^3/s.

During the RBT flow period (Apr 1 – Jun 30), flows at Brilliant Dam steadily increased and peaked on July 2, while flows at the HLK Dam were generally held stable and did not begin to increase until well into June. Peak flows from HLK Dam occurred during the third week in July; much later than is typically observed. During the flow period, flows at Brilliant Dam showed the greatest incline, escalating from approximately 700 to over 3000 m^3/s. With the record freshet in 2012, there was a greater difference between the minimum and maximum flows observed during the RBT flow period than what was previously documented (Scofield et al. 2011, Olson-Russello et al. 2012).

A downward trend of mean daily flow for both HLK and Brilliant Dams was observed during the fall fluctuating flow period (Sep 1 – Oct 31). The flows ranged from a maximum daily flow of 2078.8 and 840.2 m^3/s, to a minimum daily flow of 850.7 and 230.2 m^3/s, respectively (Table 3-2).

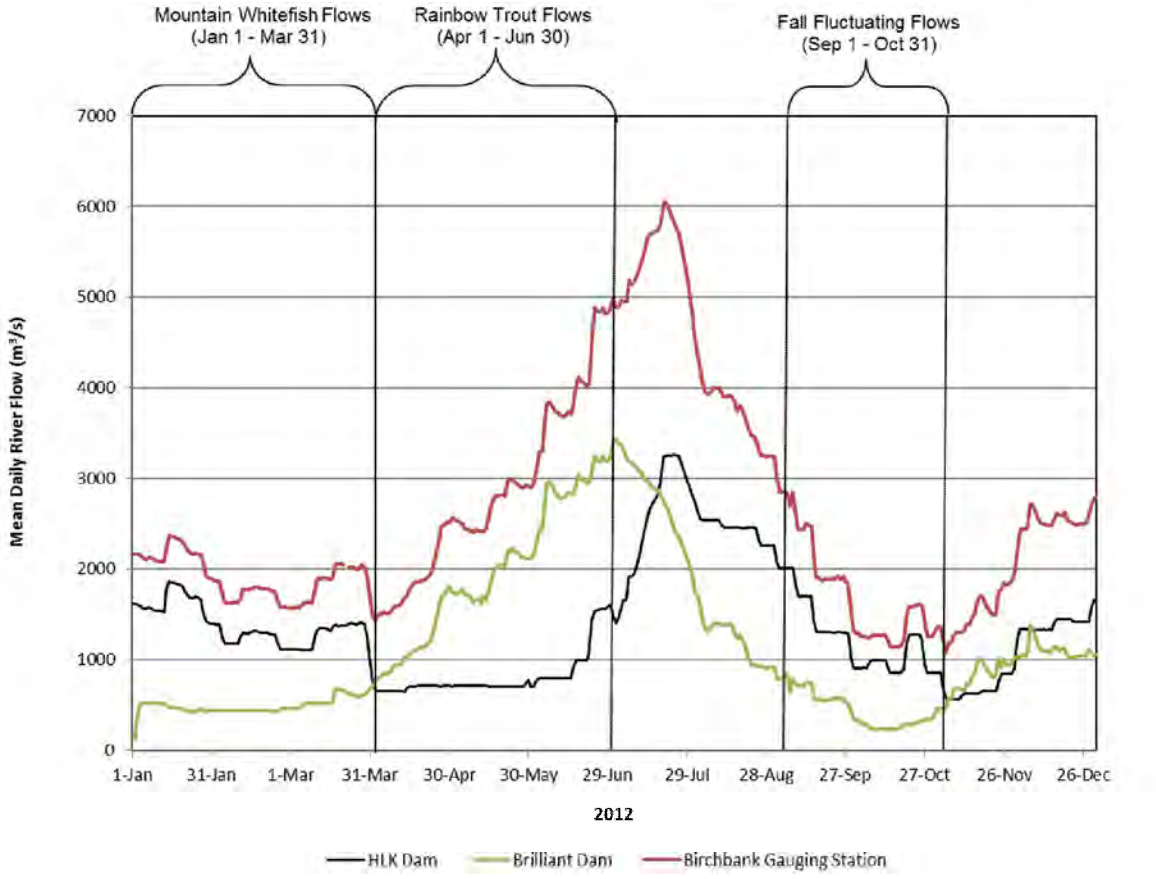

Figure 3-1: Mean Daily River Flow at HLK Dam (Columbia River), Brilliant Dam (Kootenay River), and Birchbank Gauging Station in 2012

Table 3-2: Mean Daily Flows in 2012 by Designated Flow Period (m^3/s)

Mountain Whitefish Flows (Jan 1 - Mar 31)

Year	Statistic	HLK/ALGS	Brilliant	Birchbank
	N (days)	91	91	90
	Minimum	948.6	131.2	1566.8
	Maximum	1865.3	677.5	2370.8
2012	Median	1369.8	461.1	1897.8
	Arithmetic Mean	1389.8	485.4	1913.5
	Standard Deviation	215.9	87.0	231.5
	Coefficient of Variation	0.16	0.18	0.12

Rainbow Trout Flows (April 1 to June 30)

Year	Statistic	HLK/ALGS	Brilliant	Birchbank
	N (days)	91	91	91
	Minimum	649.3	713.0	1434.8
	Maximum	1603.3	3248.0	4884.7
2012	Median	711.3	1871.9	2701.9
	Arithmetic Mean	811.0	1974.3	2841.0
	Standard Deviation	244.8	770.2	978.3
	Coefficient of Variation	0.30	0.39	0.34

Fall Fluctuating Flows (Sep 1 to Oct 31)

Year	Statistic	HLK/ALGS	Brilliant	Birchbank
	N (days)	61	61	61
	Minimum	850.7	230.2	1137.3
	Maximum	2078.8	840.2	2978.4
2012	Median	1273.2	366.0	1594.7
	Arithmetic Mean	1262.1	459.3	1764.6
	Standard Deviation	381.0	201.4	566.3
	Coefficient of Variation	0.30	0.44	0.32

3.1.2 Water Levels

In 2012, the water levels in LCR increased so substantially that all six deployed water level sensors were inundated with water, which resulted in approximately 45 days of lost data (~June 1 – Aug 15) at WQIS1 – 4. Index Station 5 (WQIS5) was the only site that did not experience lost data due to the use of a different sensor (Onset® Water Level Logger (model U20) pressure transducer) that could withstand exposure to water. However, missing data at this site did occur between May and June, as too much data had been logged since the last download and no memory space remained. Finally, the Kootenay River site (WQ C2) had the most missing data in 2012, due to lost equipment, high flows and our inability to deploy a replacement sensor due to sustained high flows.

Above average flow were observed at all sites in 2012 when compared to previous years (Figure 3-2). The Kootenay site (WQ C2) is not shown since data collection did not begin until the summer of 2011, and much of the 2012 data was lost. In 2012, recorded water level elevations above the Kootenay River confluence ranged from approximately 417.8 to

422.3 m asl. Below the confluence (WQIS4 and 5), elevations ranged from 410.4 to 417.3 m asl. The peak water level elevation was not recorded at WQIS1-3 due to lost data. Index stations 4 and 5 exhibited a higher standard deviation when compared to WQIS1-3, likely due to the influence of flows from both HLK and BRD dams.

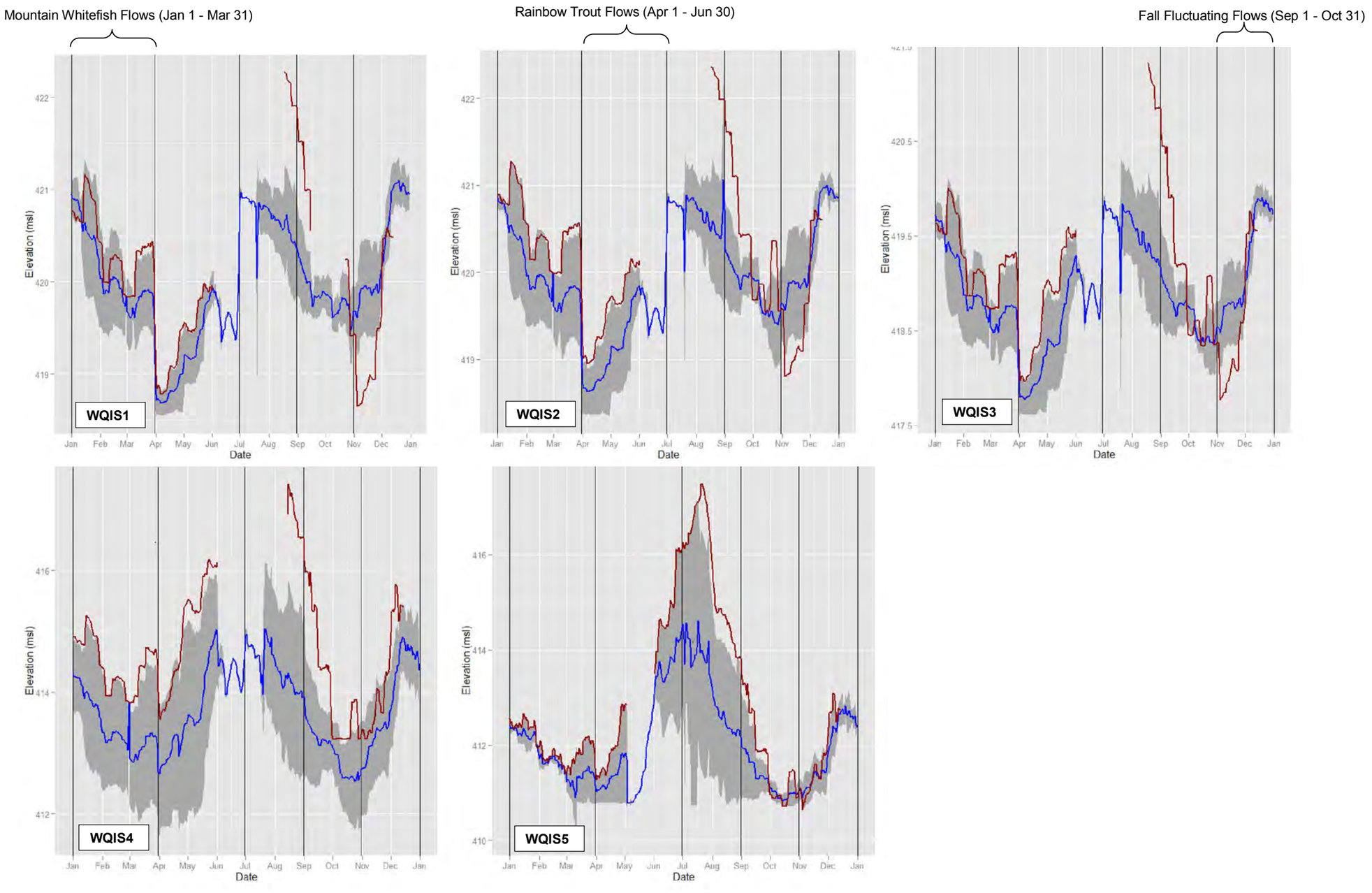

Figure 3-2: Mean daily water levels recorded at WQIS1 – 5 on LCR. The red line depicts the mean daily water level recorded at each site in 2012. The blue line is the mean daily water level throughout the duration of the study (2008-12) ± SD (gray shaded area). The SD could not be determined for all months due to gaps in data collection.

3.1.2.1 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), a model of flow and elevation was generated independently for each WQIS. The "best fit model" for each site was determined using model averaging to select the most appropriate explanatory variables relating elevation and flow. Table 3-3 outlines all possible flows and their associated quadratic terms that were used in determining the best fit model. The squared terms were included to help account for curved relationship between flow and elevation. The best fit models varied among the five sites, and was dependent on where a site was located in relation to source flows. Sites above the BRD confluence were best predicted by flows from HLK and sites downstream were more dependent upon flows measured at BBK or BRD dam (Table 3-6). The R^2 values for the top models ranged from 0.86 to 0.98, and all models had highly significant p-values ($p<0.001$).

Table 3-3: Possible flows used in regression modeling for predicting water levels during the MWF flow period

Possible Predictor Flows
HLK flow
HLK flow + HLK flow ²
Brilliant flow
Brilliant flow + Brilliant flow ²
Birchbank flow
Birchbank flow + Birchbank flow ²

Historic river elevation data was not available, so the predicted elevations generated using the best fit models were used to analyze historic river elevations during periods before implementation of MWF (pre), prior to the study period during the initial implementation of MWF (post), and during the study period (continuous). The difference in river level elevation between the maximum peak spawning flow and the minimum incubation flow for each year was determined for each period (pre (1984 – 1994), post (1995 – 2007) and continued (2008-2012)) in the analysis.

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. 2, 26, $p<0.001$). Figure 3-3 also depicts the actual elevation difference compared to the elevation from predicted models from the continuous flow period, confirming that selected models are good predictors of historic river elevation.

Based on these analyses, it appears 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. Scofield et al. (2011) reported similar findings.

Table 3-4: The "best fit models" for each WQIS that were used to predict historic water levels during the MWF flow period. Standard error for terms were obtained using model averaging.

	Site	Best Fit Model (Intercept +Coefficient (± SE))	Adjusted R^2	p-value
MWF Analysis	WQIS1	417.5 + HLK (.0029 ± . 8.29e-04) + HLK^2 (-4.847e-07 ± . 2.377708e-07)	0.94	<.0001
	WQIS2	415.4 + BBK (.0044 ± .0011)+ BBK^2 (-1.28e-06 ± 4.46e-07) + HLK (-0.00021 ± 1.017e-03) + HLK^2 (8.001e-07 ± . 3.68e-07)	0.98	<.0001
	WQIS3	416.6 + BBK (0.00039 ±. 8.46e-04) + HLK (0.0018 ± 5.73e-04) + HLK^2 (-2.352e-07 ± 1.48e-07)	0.97	<.0001
	WQIS4	409.1 + BBK (0.0026 ± 1.49e-03)	0.86	<.0001
	WQIS5	407.9 + BBK (0.0027 ± . 6.15e-04)+ BBK^2 (-2.96e-07 ± 1.67e-07) + BRD (0.00062 ± 4.29e-04) + BRD^2 (-1.32e-06 ± 6.53e-07)	0.95	<.0001

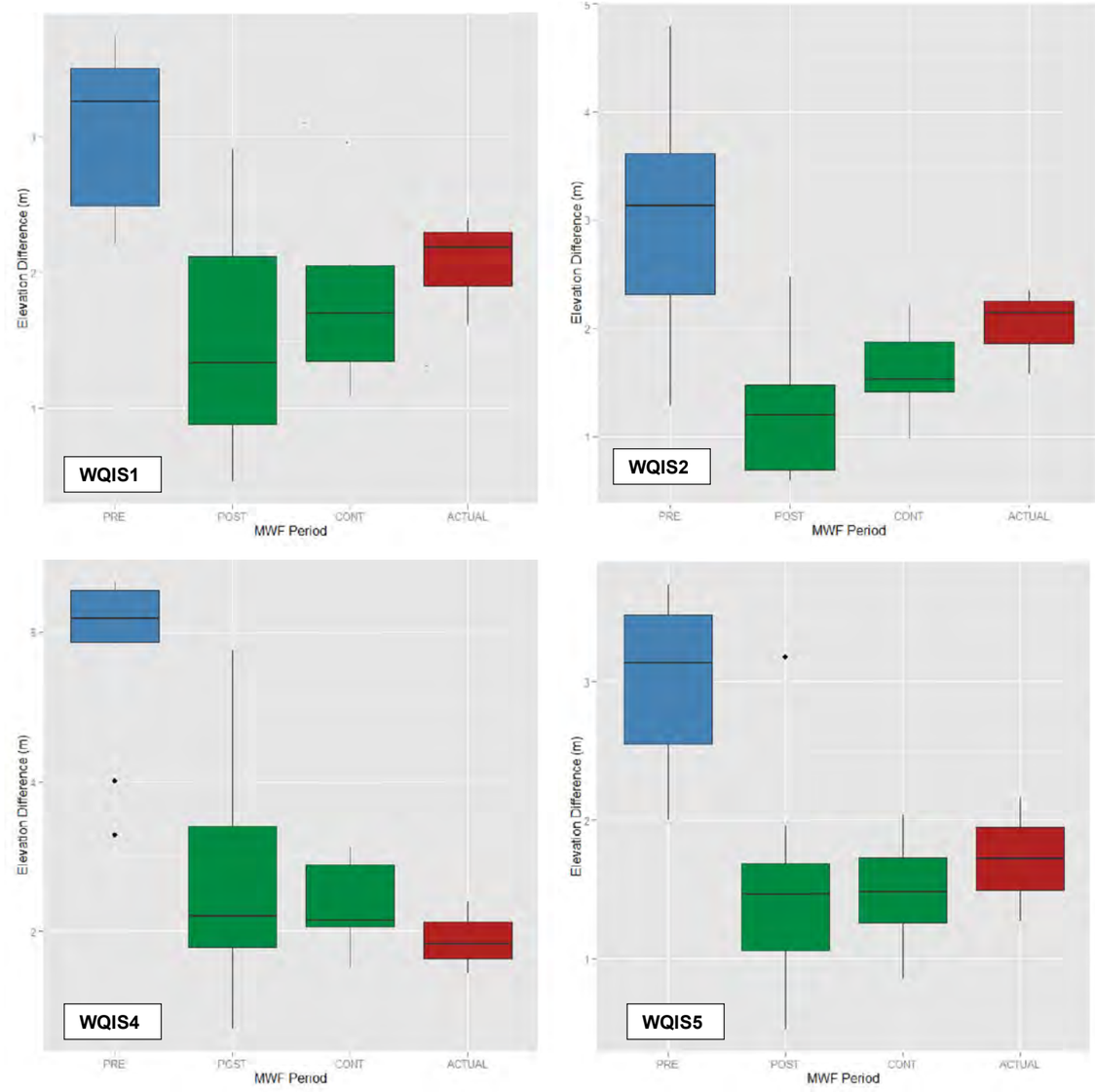

Figure 3-3: Predicted water level elevation difference between maximum flows during Mountain Whitefish (MWF) spawning (Jan 1 – Jan 21) and minimum flows during MWF egg incubation (Jan 22 – Mar 31) for Pre (1984 – 1994), Post (1995-2007), and Continuous (2008-2012) flow years at each water quality index station. Different colours indicate statistical significance ($p<0.05$) as determined by a permutation ANOVA. The "actual" dataset was not statistically analyzed but is included to illustrate variability between predicted CONT values and actual elevation field data collected during 2008-2012.

3.1.2.2 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 1 April and 30 June, we used the same analysis procedure described above for sub-hypothesis $HO_{2Aphy.}$ To limit the analysis to the Norns Creek fan, regression modeling was only undertaken for the WQIS2 and 3, which are closest. The same possible predictive flows outlined for the MWF analysis were used to determine the best fit model (Table 3-5). The best fit model varied for the two sites, with flows from BBK, BRD and HLK all having a positive effect for WQIS2 during the RBT flow period. The best fit model for WQIS3 only consisted of flows from BBK and HLK (Figure 3-4).

Similar to the MWF analysis, the best fit model was used to predict historic flows for each site during each time period (pre, post, and continuous). Drops in elevation were added across the entire flow period to determine the total effective elevation drop to address the management question.

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 continued (2008-2012) flow periods (perm. ANOVA: WQIS2, d.f. 2, 26, $p<0.001$; WQIS3, d.f. 2, 26, $p=.0028$). Similar to the MWF flow period, predicted elevations were consistent with field observations of river elevation drops observed at WQIS2 and 3.

Table 3-5: Best fit models for WQIS2 and 3 that were used to predict historic water levels during the RBT flow period

	Site	Best Fit Model (Coefficient ± SE)	Adjusted R^2	p-value
RBT Analysis	WQIS2	417.2 + BBK (2.11e-04 ± 1.217e-04) + BRD (4.65e-05 ± 9.78e-05) + BRD^2 (1.56e-07 ± 5.96e-08)+ HLK (0.0021 ± 3.62e-04)	0.95	<.0001
	WQIS3	416.9 + BBK (2.002e-04 ± 9.65e-05) + BBK^2 (1.70e-07± 2.92e-08)+ HLK (0.00085 ± 3.386638e-04)	0.97	<.0001

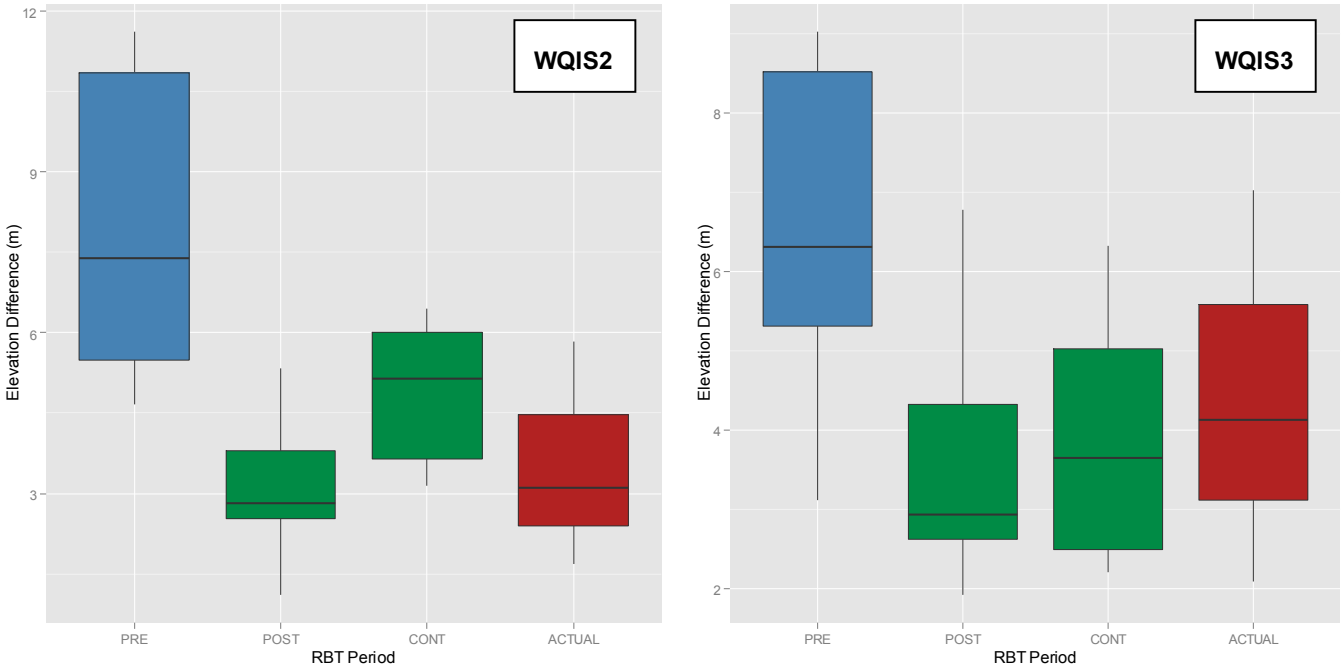

Figure 3-4: Cumulative sum of elevation drops occurring during the Rainbow Trout Flow period for Pre (1984 – 1991), Post (1992-2007), and Continuous (2008-2012) flow years at each water quality index station. Different colors within each graph for Pre, Post and Cont datasets indicate statistical significance (p<0.05) as determined by a permutation ANOVA. The "Actual" dataset was not statistically analyzed but is included to illustrate variability between predicted CONT values and actual elevation field data collected during 2008-2012.

3.2 Physical and Chemical Characteristics

3.2.1 Water Temperature

As with the flow data, 2012 water temperature data also had data gaps, most notably during the summer high flow period (Figure 3-5). Water temperatures during 2012 at the five WQIS varied seasonally, ranging from approximately 4 to 18°C. The 2012 summer daily temperatures were several degrees lower than the mean temperatures recorded during previous years of the study. The unusually high flows of 2012 probably contributed to these lower temperatures. The temperature discrepancy between years diminished by September (see Figure 3-5; WQIS5).

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) 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.

The vertical lines on Figure 3-5 indicate the beginning and end of each flow period. During MWF flows (Jan 1 – Mar 31), the 2012 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 5 to 12 °C. Finally, the fall fluctuating flow

period exhibited the opposite trend with water temperatures declining from approximately 16 to 10 °C.

To address hypothesis Ho_{1phy}, that states 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, a linear mixed effect model was used, with Year and Season considered as random effects. To ensure the model was representative of variable site conditions above and below the confluence of the Kootenay River, explanatory variables were weighted by flows originating from either HLK or BRD. Explanatory variables considered in the analysis were Castlegar air temperature (castle_temp), flow period (Mountain Whitefish (FPMWT), Rainbow Trout (FPRBT)), Summer (FPSUM), and Winter (FPWIN), the weighted averages of Arrow Lakes (HLK) and Kootenay Reservoir (BRD) elevations (Res.Temp), the weighted elevation of Arrow Lakes (HLK) and Kootenay Reservoir (BRD) (Elev), and weighted flows originating from both HLK and BRD (Flow) at each site. Explanatory variables were standardized to ensure they could be compared directly.

Model averaging was used to determine plausible models. There was only one model with an ΔAICc <2 (R^2 = 0.92); this model included all explanatory variables and interactions of flow period with all explanatory variables. The model explained a very high proportion of the variance (R^2 = 0.93). Castlegar air temperature and reservoir temperature were most strongly correlated with water temperatures when all seasons and flow periods were considered, with the greatest effects observed in the fall, winter, and rainbow trout periods (Figure 3-6).

Reservoir temperature was considered the next most important determinant of river temperature, with the greatest effects observed during fall, winter, and mountain whitefish flow periods where temperature decreased with increasing reservoir elevation. The effect of reservoir elevation was greatest during the summer period, with river temperature decreasing with increasing reservoir elevation, indicating that water released into LCR from deeper in the reservoir is cooler during the summer. The effects of flow on river temperature was positively associated with flow during the fall and rainbow trout flow period, and negatively associated with flow during the summer and winter. River flows appeared to have no effect during the mountain whitefish period.

Based on this preliminary analysis, water temperature may be influenced by flow, but flow is not the most important determinant of river temperature. The effects of flows on river temperature were greatest during winter and fall, with a marginal effect of increasing river temperature with increasing flows observed during the rainbow trout flow period.

Table 3-6: Coefficients and Standard Errors of multiple explanatory variables used to predict LCR water temperature that were obtained using linear mixed effects models average across all models with an AICc < 2. Year (2008-2012) and Site (WQIS1 through WQIS5) were treated as random effects within the model. Definitions for terms are in the table footnote.

Variable	Coefficient	Std. Error	P Value
(Intercept)	-3.68E+01	9.03E+00	<0.001
castle_temp	3.63E-01	2.48E-02	<0.001
Elev	3.63E-01	2.07E-02	<0.001
Flow	1.06E-03	3.44E-04	0.002
FPMWT	4.99E+01	1.38E+01	<0.001
FPRBT	3.84E+01	1.14E+01	<0.001
FPSUM	1.08E+02	1.58E+01	<0.001
FPWIN	2.41E+01	1.76E+00	0.17
Res.Temp	-1.48E-01	1.07E-01	0.16
castle_temp:FPMWT	-3.04E-01	4.09E-02	<0.001
castle_temp:FPRBT	-9.43E-03	3.93E-02	0.81
castle_temp:FPSUM	-2.79E-01	4.60E-02	<0.001
castle_temp:FPWIN	-2.74E-01	4.92E-02	<0.001
Elev:FPMWT	-1.38E-01	3.19E-02	<0.001
Elev:FPRBT	-1.13E-01	2.65E-02	<0.001
Elev:FPSUM	-2.37E-01	3.56E-02	<0.001
Elev:FPWIN	-7.13E-02	4.07E-02	0.08
Flow:FPMWT	-1.64E-03	4.83E+00	<0.001
Flow:FPRBT	-2.05E-03	4.18E-04	<0.001
Flow:FPSUM	-1.65E-03	3.97E-04	<0.001
Flow:FPWIN	-2.41E-03	4.89E-04	<0.001
FPMWT:Res.Temp	1.10E+00	1.87E+00	<0.001
FPRBT:Res.Temp	9.58E-01	1.67E-01	<0.001
FPSUM:Res.Temp	1.99E-01	1.83E-01	0.28
FPWIN:Res.Temp	8.52E-01	1.74E-01	<0.001

castle_temp = Castlegar Air Temperature, Elev = Reservoir Elevation, Flow = Standardized flow, FPMWT = Mountain Whitefish Flow Period, FPRBT = Rainbow Trout Flow Period, MWTSUM = Summer Flow Period, FPWIN = Winter Flow Period, Res.Temp = Standardized Reservoir Temperature.

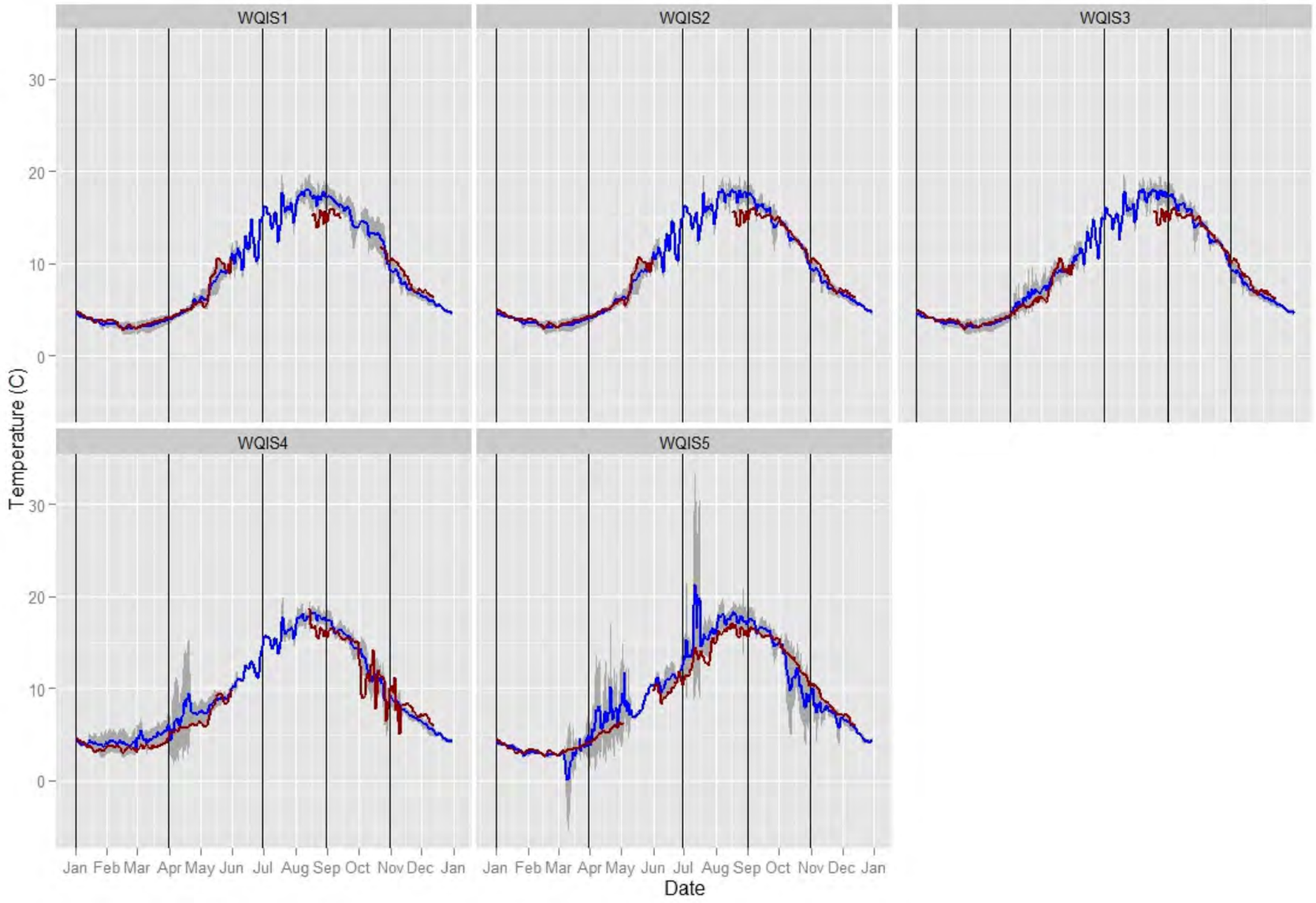

Figure 3-5: Mean daily water temperatures recorded at WQIS1 – 5 on LCR. The red line depicts the mean daily water temperature recorded at each site in 2012. The blue line is the mean daily water temperature throughout the duration of the study (2008-12) ± SD (gray shaded area). The vertical lines indicate the beginning and end of each flow period. MWF flows occurred between Jan 1 and Mar 31, RBT flows occurred between Apr 1 and Jun 30 and fall fluctuating flows occurred from Sep 1 to Oct 31.

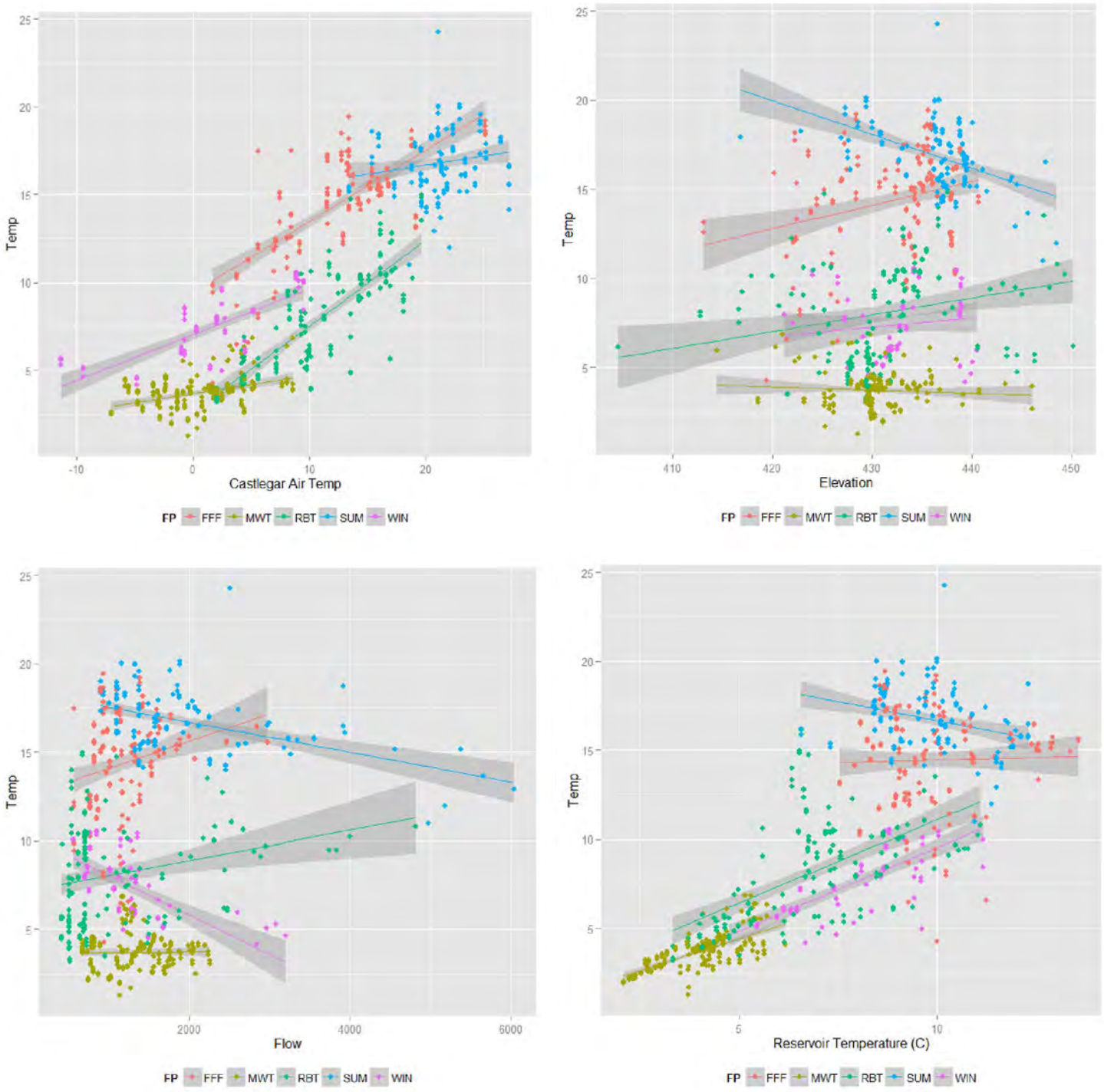

Figure 3-6: Single linear regressions of water temperature data and Castlegar air temperature, weighted reservoir elevation, weighted LCR flow, and weighted reservoir temperature. Flow period (FFF = Fall fluctuating flows, MWT = Mountain Whitefish flows, RBT = Rainbow Trout flows, SUM = Summer flows, and WIN = Winter Flows) interactions were significant and a separate regression for each flow period is presented.

3.2.2 Water Quality

Water quality sampling was modified from the previously collected monthly samples in the June to October growing season to allow sampling to be more disbursed annually and to achieve an overlap with the MWF flow period. Samples were collected on June 1, August 14, and October 25 2012, and will be collected near March 30, June 1, September 1, and November 1 in 2013.

Appendices A1 and A2 contains all 2012 raw water quality data, including field and laboratory results.

3.2.2.1 pH

During 2012, mean pH was 7.76 ± 0.14 (SD) and ranged from 7.53 – 8.04, with the highest values recorded in August (Figure 3-7). The pH in Kootenay River ranged from 7.56 – 8.15 and the mean pH was 7.78 ± 0.32. pH readings from 2012 were very similar to those reported during the first four years of this study (Scofield et al. 2011, Olson-Russello et al, 2012). Of LCR sites, the highest average pH occurred at WQSI5. All LCR pH values met the BC MoE Guideline and fell within LCR Objective range.

The pH of Norns Creek was lower than LCR in 2011, but higher in 2012, ranging from 7.77 to 8.60. The change in sampling frequency, with two readings during extreme low flows increased the 2012 average pH to 8.23 ± 0.42. The August pH reading taken from Norns Creek during low, warm flows was slightly higher than the upper LCR objective limit.

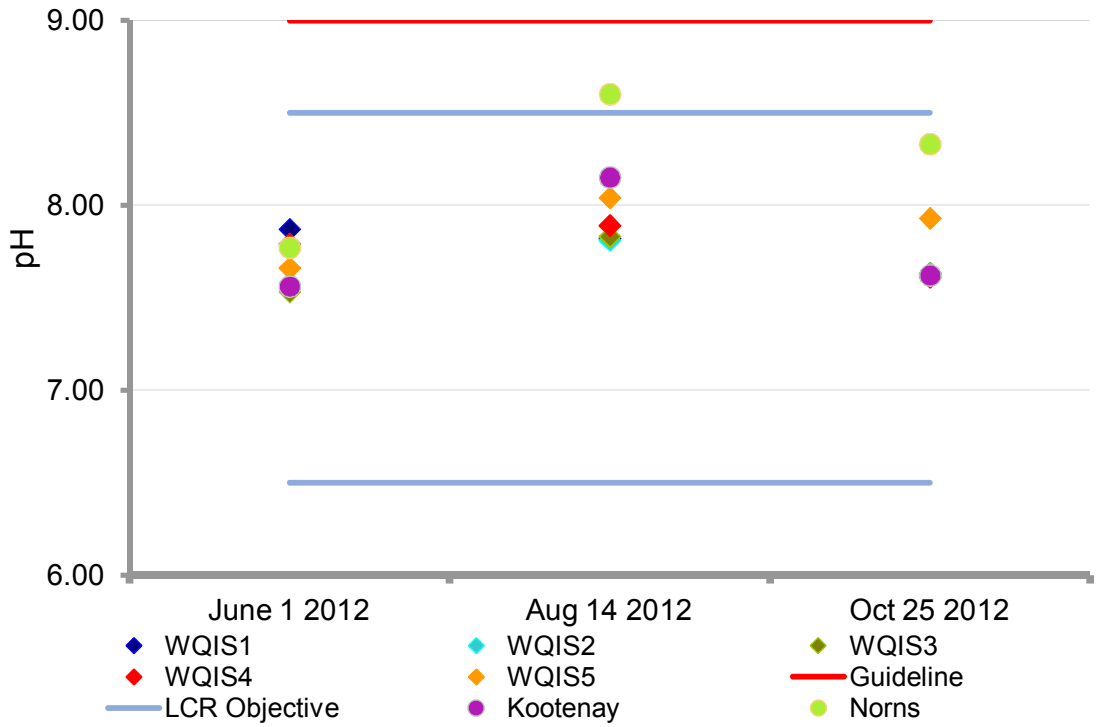

Figure 3-7: pH from LCR Water Quality Index Sites and Main Tributary Sites during 2012

3.2.2.2 Electrochemistry Parameters

Specific conductance, total dissolved solids (TDS), alkalinity and hardness all measure the concentrations of ionized constituents in water and frequently trend together (Table 3-7). There is some overlap in the measured ions. For example, measurements 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 3-7: Ions Contributing to Electochemistry Parameters

Parameters	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 CI-S04-2 N03- HCO-, OH-

In both LCR and its tributaries, specific conductance showed an inverse relationship with flow. Conductivity during freshet was half of the conductivity during the low flow period. Specific conductance ranged from 50 – 116 µS/cm in LCR during 2012 (Figure 3-8), which was lower than the range reported for these sites during 2008 - 2010 (82 – 148 µS/cm). The lower specific conductance observed in both 2011 and 2012 is probably the result of dilution during these record freshet years. The range of specific conductance measured at Birchbank between 1983 and 1996 was higher (105 – 160 µS/cm) than the range reported here (Holmes and Pommen 1999).

Kootenay River had consistently higher specific conductance measurements than LCR during 2012. It averaged 105 ± 32 µS/cm compared to 92 ±2 4 µS/cm in LCR samples. Like 2011, Kootenay River samples ranged from 67 – 126 µS/cm and these values were noticeably lower than the range (105 – 149 µS/cm) reported for 2008 – 2010 (Scofield et al. 2011). Norns Creek values ranged from a very low 9 to 96 uS/cm, consistent with historic values. The low conductance observed at Norns Creek is typical of streams whose source is mostly snowmelt.

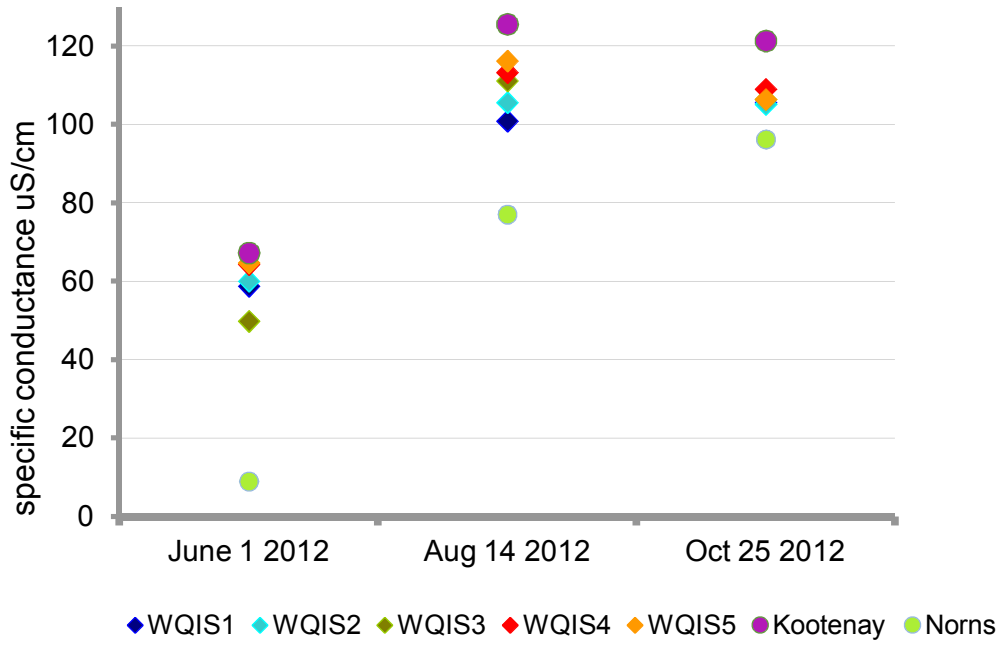

Figure 3-8: Specific Conductance from LCR Water Quality Index Sites and Main Tributary Sites during 2012 (no guideline or objective available)

Total dissolved solids results from the lab (not field meter TDS results) are shown in Figure 3-9. TDS averaged 58 ± 10 in LCR mainstem sites during 2012. The 2012 results were lower than those from 2011, particularly during the record freshet. Like conductivity, TDS tended to increase as the water travelled through LCR with the exception of October's results that showed no pattern. As in previous years, TDS in Kootenay River exceeded that of LCR, averaging 70 ± 17 mg/L TDS during 2012. The higher TDS observed in Kootenay River is reflected in observed increases in TDS in LCR downstream of the confluence.

Norns Creek always had low conductivity and TDS, even during very low flows. This suggests that the watershed consists of mostly granite (non-carbonate) or some other hard, non-ionic forming type of rock.

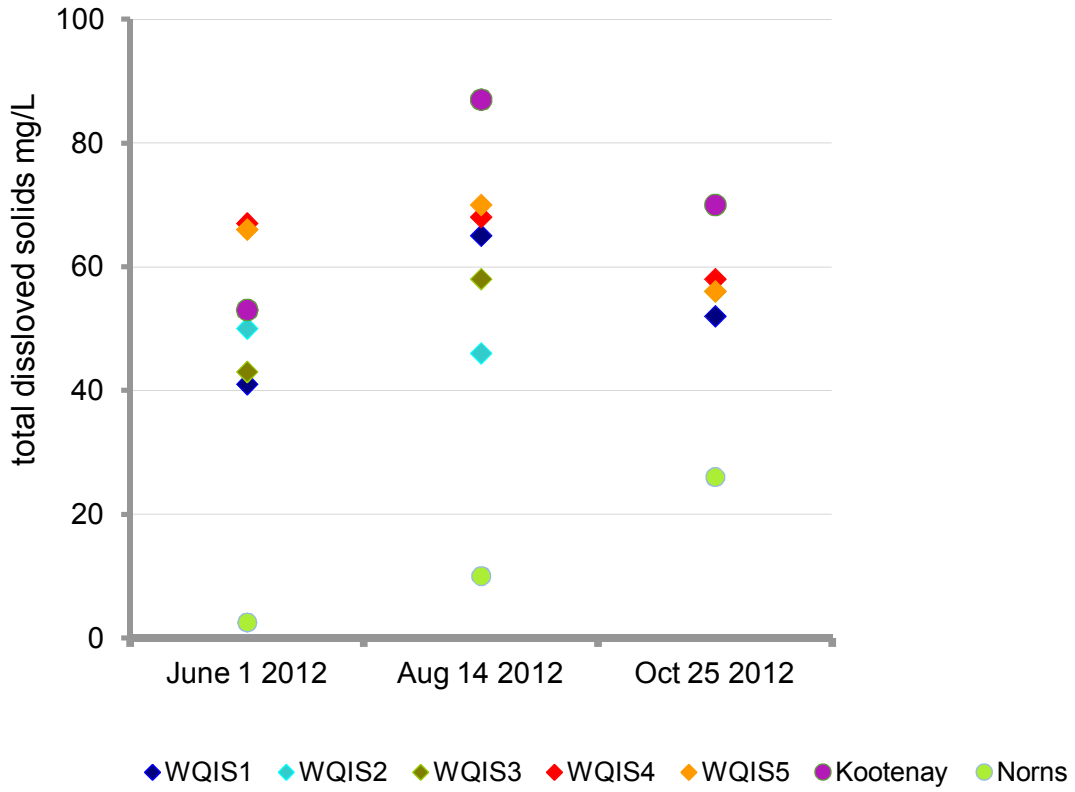

Figure 3-9: Total Dissolved Solids from LCR Water Quality Index Sites and Main Tributary Sites during 2012 (no guideline or objective available)

Like specific conductance and TDS, hardness was low and stable in LCR, ranging from 50 – 70 mg/L during the summer flow period (Appendices A3 & A4). Similarly, alkalinity ranged from 45 – 52 mg/L in 2012 LCR samples, while the Kootenay River measured 61 – 64 mg/L. The highest alkalinity in LCR was found at WQ S5, below the confluence with the Kootenay. The higher alkalinity of the Kootenay tends to buffer pH fluctuations, while the lower alkalinity of LCR would cause larger changes in pH with an addition of low pH water.

3.2.2.3 Inorganic Nitrogen

Nitrate concentrations averaged 0.094 ± 0.01 mg/L, while ammonia and nitrite were consistently non-detectable in the 2012 LCR samples. These nitrate concentrations were significantly higher than the average reported for 2008-2010 of 0.051 mg/L (Scofield et al. 2011). 2012 freshet nitrogen concentrations were comparable to 2011, but the later sample dates were elevated, and may relate to the fertilization program on the Arrow Lakes Reservoir. During the period when added nitrogen should arrive at LCR (July through October) inorganic nitrogen concentrations were highest at the sites closest to the dam, and diminished as the water flowed downstream, despite two municipal outfalls and Celgar. The 2012 average was affected by the altered sampling frequency compared to the earlier years of this study.

During 2012, the Kootenay River samples averaged 0.069 ± 0.03 mg/L nitrate as N, and like 2011, these concentrations were lower than LCR. In 2011 and 2012, the Kootenay had similar nitrate concentrations to LCR during freshet, but dropped lower during clear flow period (Figure 3-10). Both years were elevated compared to 2008-2010 (Scofield et al. 2011). Nitrate concentrations were much lower in Norns Ck than the rivers, averaging 0.013 ± 0.005 mg/L as N. It is the site with consistently low nitrates but moderate phosphorus. Agriculture occurs along Norn's lower length, but does not appear to result in elevated inorganic nitrogen concentrations.

All of LCR sites, the Kootenay and Norn's Ck were far below the BCMOE aquatic life nitrogen guidelines of 3 mg/L nitrate and 0.7 mg/L ammonia.

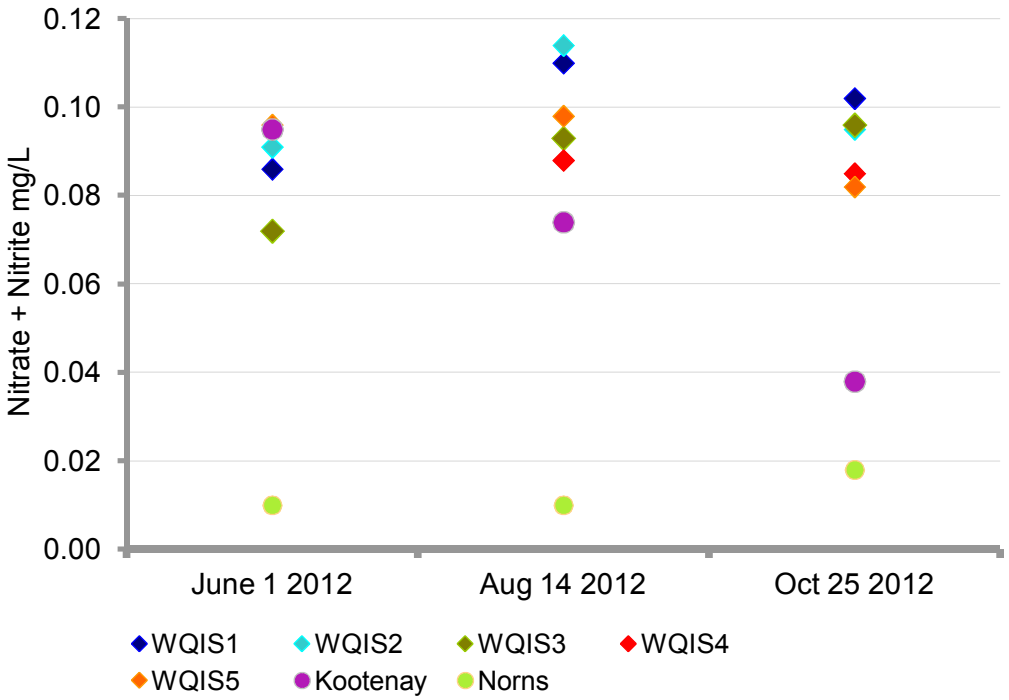

Figure 3-10: Nitrate and Nitrite from LCR Water Quality Index Sites and Main Tributary Sites during 2013 (BCMOE guideline is 3 mg/L nitrate; 0.7 mg/L ammonia)

3.2.2.4 Phosphorus

Total Phosphorus (TP) is the total dissolved plus particulate phosphorus in a water sample. In addition to biologically available P, total phosphorus can include organic phosphates, P-bearing minerals and P absorbed onto to mixed phases (e.g. clays, organic complexes, metal oxides and hydroxides) (Maher and Woo 1998).

Total phosphorus (T-P) concentrations in LCR were relatively low in all 2012 samples (Figure 3-11). Unlike 2011, T-P concentrations in LCR and Kootenay were very low during the 2012 freshet at <0.002 mg/L. The 2012 average T-P concentration in LCR was 0.006±0.005 mg/L. Some uncertainty is introduced to these averages by the frequent results at or below the reportable detection limits of the lab analyses. Further, the reduction from five sample dates to three reduces the certainty of the 2012 averages.

Inorganic ortho-phosphate (or SRP) represents the fraction of T-P that is available to organisms for growth. In LCR, it never exceeded the detection limit of 0.01 mg/L as P in 2011 or 2012 samples, with one exception – WQIS4, which is downstream of the Kootenay confluence and several outfalls. One City of Castlegar outfall is located at WQIS3, but with the composite sampling across the river transect, these samples did not have detectable ortho-phosphate concentrations. The Kootenay samples averaged 0.011 ± 0.009 mg/L T-P during 2012 and 0.009 ± 0.01 mg/L T-P during 2011, both higher than the 0.005 mg/L T-P measured during 2008 – 2010 (Scofield et al. 2011), however non-detectable results were common and reduce the accuracy of these means. Overall, the 2012 T-P concentrations would classify the Kootenay River as mesotrophic. The Norns Creek samples averaged a modest 0.015 ± 0.009 mg/L during 2012, which was higher than the 2011 results. T-P frequently trends with flow in the smaller LCR tributaries.

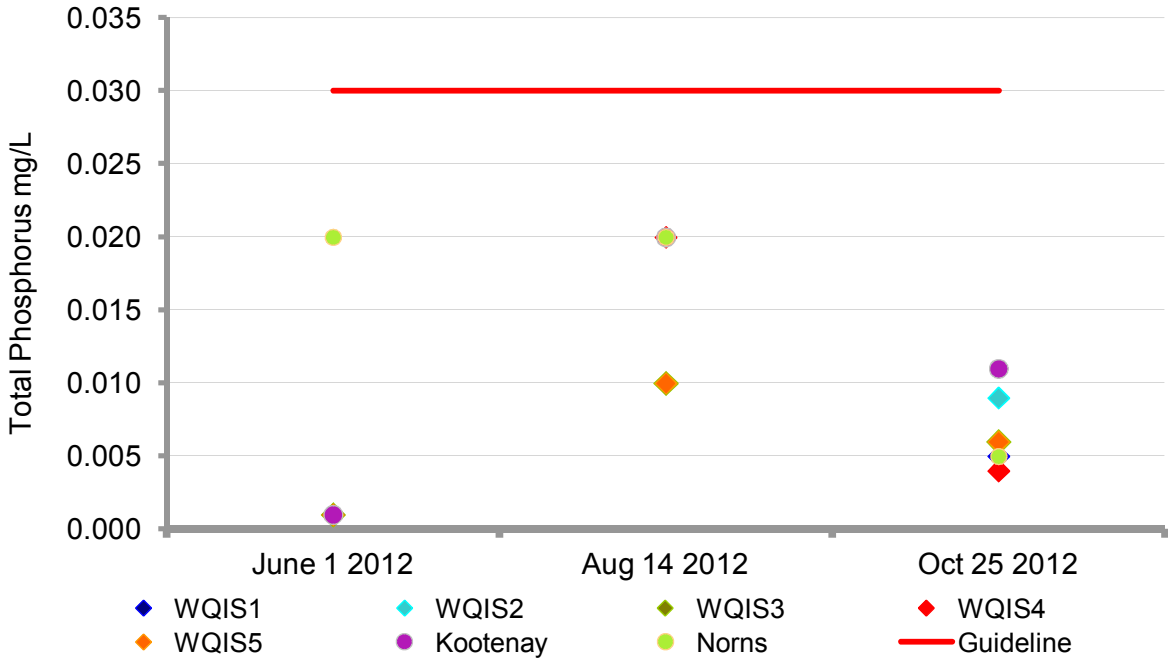

Figure 3-11: Total Phosphorus from LCR Water Quality Index Sites and Tributary Sites during 2012 (tentative guideline = 0.03 mg/L T-P)

3.2.2.5 Turbidity

In LCR, turbidity is typically low. In both 2011 and 2012, turbidity ranged from 0.3 to 0.9 NTU with an average of 0.50±0.17 NTU, all within the range reported during 2008 – 2010 (Scofield et al. 2011). No turbidity spikes equating to the magnitude seen in past years of >7 NTU were observed in 2011 or in 2012 (Figure 3-12). Turbidity at sites WQIS4 and WQIS5 below the confluence with Kootenay River had slightly higher turbidity than the sites above the confluence.

As in most years, the Kootenay had higher turbidity than LCR in 2012. The record high flows in the 2011 and 2012 freshets increased mean and peak turbidity in the Kootenay River to 0.70 and 0.80 NTU, respectively. Both the Kootenay and LCR are fed from reservoirs that allow settling of the suspended materials that cause turbidity. Turbidity in Norns Ck measured 4.1 NTU during freshet because it is run-of-river, without a reservoir. Both LCR and the Kootenay met all turbidity criteria in the 2012 samples.

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 have hindered photosynthesis (Caux et al. 1997; ENSR 2001). However, light penetration

through deeper water in the mid-depth and beyond would be reduced enough to influence periphyton production (Appendices A3 and A4**).**

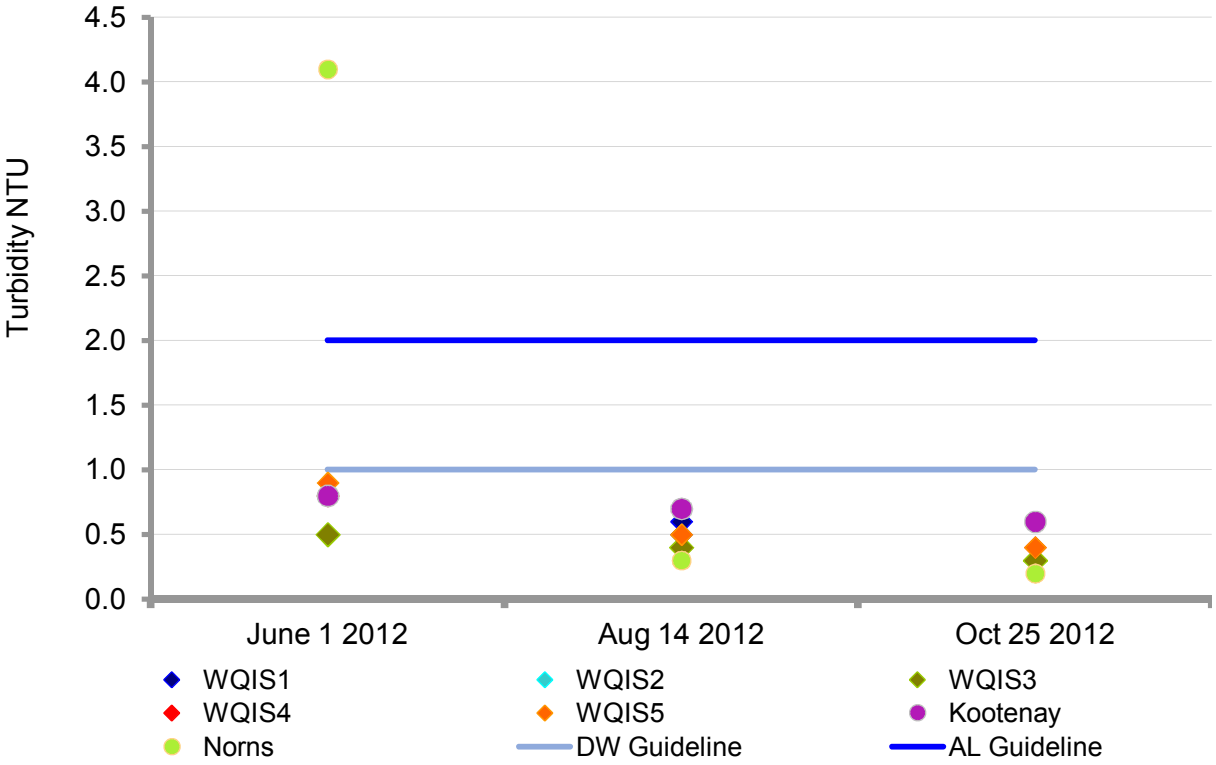

Figure 3-12: Turbidity from LCR Water Quality Index Sites and Main Tributary Sites during 2012 (aquatic life protection guidelines state maximum 24 hr increase = 8 NTU; maximum clear flow average (30 day) increase = 2 NTU; drinking water guideline =1 NTU)

3.2.2.6 Total Suspended Solids

Total suspended solids concentrations are typically low in both LCR and Kootenay Rivers, and were low again in 2012, ranging from <1 - 2 mg/L. Total suspended solids in LCR was usually less than 1 mg/L, however, TSS measured 2 mg/L in Kootenay and the downstream LCR sites (WQIS4 and WQSI5) during August (Figure 3-13). The very large diatoms that grow in Kootenay Reservoir may have contributed to this observed TSS increase. The 2012 Norns Ck freshet had sufficient velocity on June 2 to raise TSS to 15 mg/L. During low flows, Norns Ck remained below the TSS detection limit of <1 mg/L

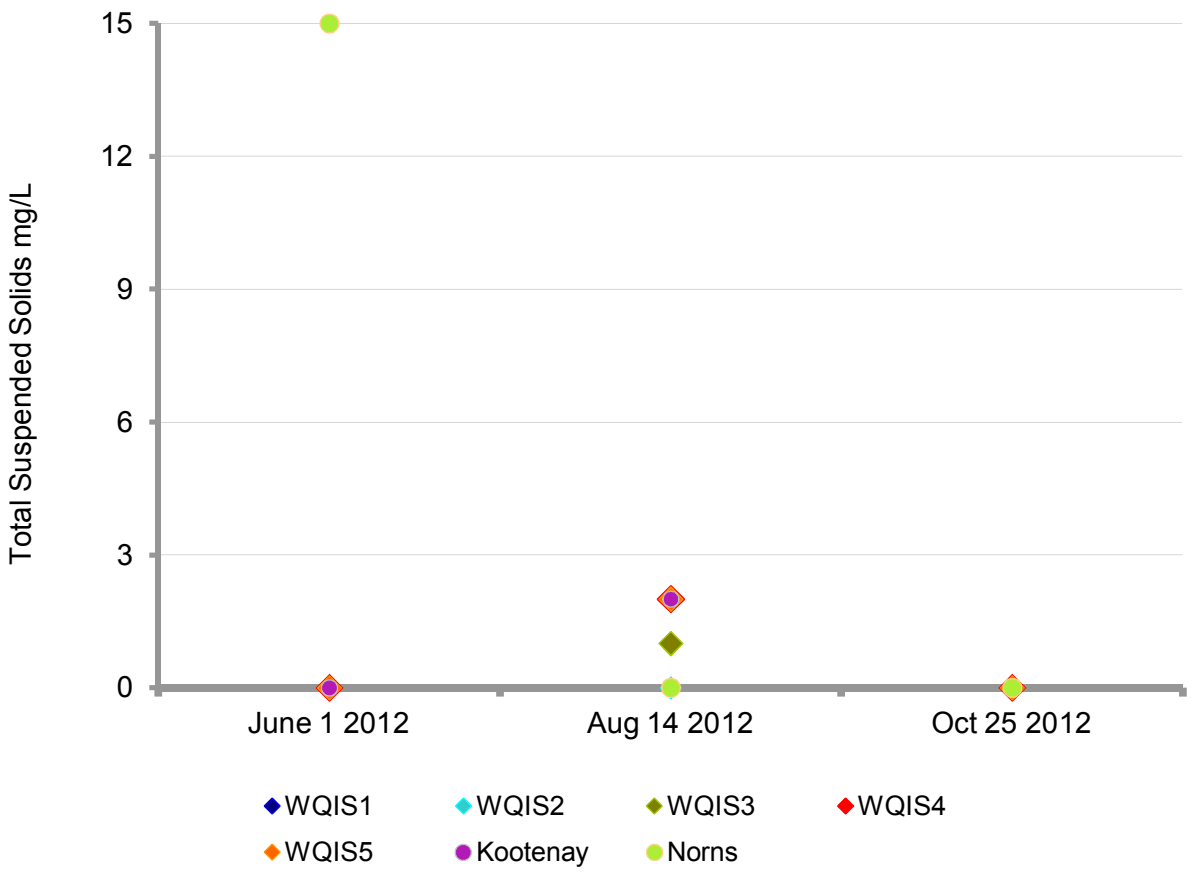

Figure 3-13: Total Suspended Solids from LCR Water Quality Index Sites and Main Tributary Sites during 2012

3.2.2.7 Dissolved Oxygen

All LCR sites met the BCMoE guideline while a few measurements were lower than LCR objective set by the Province in both 2011 and 2012. Dissolved oxygen ranged from 9.7 – 12.6 mg/L from June to October in LCR, a range very similar to previous years, despite the change in sampling frequency. Dissolved oxygen declined over the summer in response to increased water temperature. LCR sites averaged 10.6 ± 1.1 (SD) mg/L DO, unchanged from 2011 (Figure 3-14). Dissolved oxygen saturation ranged from a minimum of 95% in October to a maximum of 120% on a sunny day in June. During 2012, the average DO saturation was 107 ± 9%. The sites with the lowest DO saturation readings in 2012 were immediately downstream of HLK Dam (WQIS1 and WQIS2), and dissolved oxygen generally increased as the water travelled downstream. Dissolved oxygen concentrations have been adequate for all salmonid life stages throughout this study (BC MoE 2012).

Dissolved oxygen in the Kootenay River ranged from 9.6 – 13.2 mg/L, slightly higher than the 9.3 – 12.2 mg/L reported from 2008 – 2010 (Scofield et al. 2011), and slightly higher than LCR. The average DO at the Kootenay River site was 10.8 ± 2.1 (SD) mg/L or 110% saturation during 2012. Like LCR, Kootenay River met the BCMOE guideline while a few measurements were lower than LCR objective set by the Province. After Kootenay River, Norns Creek is the largest tributary to LCR and it measured 9.0 – 13.2 mg/L DO, a range within that reported previously (Scofield et al. 2011). Readings were taken within 1 m of the substrate in Norns Creek and averaged 101% oxygen saturation during daylight hours. The August Norns Ck dissolved oxygen sample was at the guideline as a result of low, warm flows.

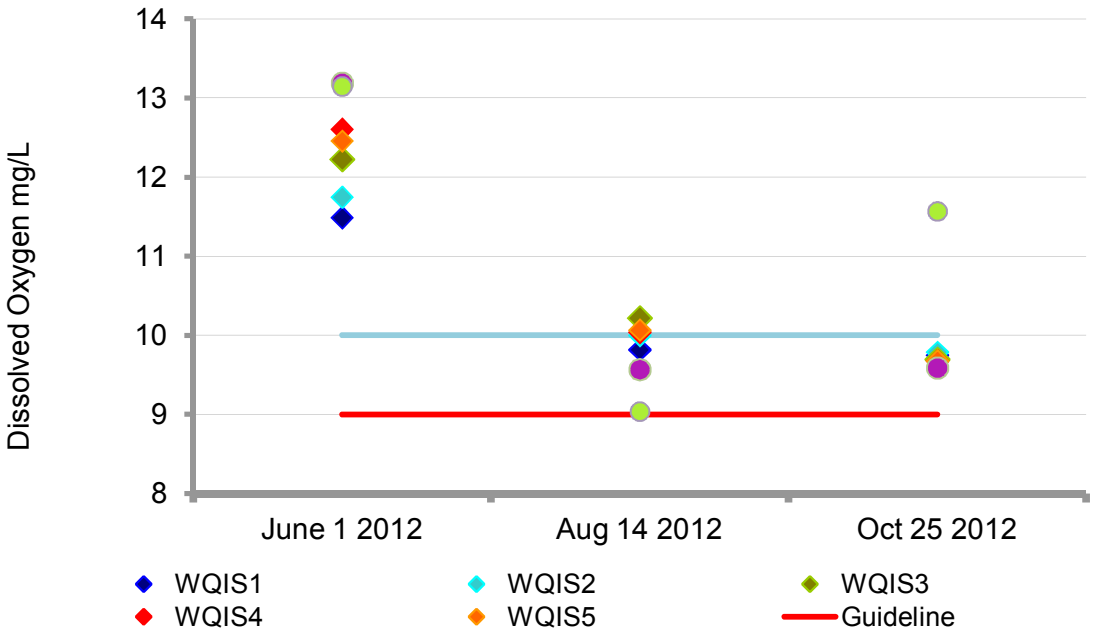

Figure 3-14: Dissolved Oxygen mg/L from LCR Water Quality Index Sites and Main Tributary Sites during 2012

3.2.2.8 Water Quality Hypothesis Testing

Statistical analyses to address water quality hypotheses will be included in future years of the study once more data is available from fish flow periods. As a first step, we have used principal component analysis for exploratory data analysis, with the intent of determining if there is a relationship between flow and any of the electrochemistry and/or biological active nutrient parameters. The outputs of this analysis thus far have been difficult to interpret. Further work is needed to hone in on the most appropriate analysis. If relationships are revealed, then we should be able to subsequently understand how flows during specific fish flow periods may or may not affect the water quality of LCR.

3.3 Periphyton

3.3.1 Composition and Abundance of Biofilm Bacteria and Fungi

Periphyton consists of two groups of micro-organisms, photosynthetic bacteria and algae, plus heterotrophic bacteria and fungi. Algal periphyton production can only occur while substrates are submerged and in the light, while the bacterial biofilm component also grows in the dark (Lear *et al.* 2009). Viable periphyton found in LCR can be further subdivided into in situ periphyton utilizing locally available nutrients, and algae exported by the upstream reservoirs that remain viable after they adhere to the substrate biofilm. Based on the drift samples, significant additional food for benthic invertebrates is imported to LCR as leaves, seeds, pollen and soil detritus. Together these sources provide the basis for LCR food chain.

Bacteria and fungi (moulds, yeasts) are pioneering organisms that can dominate the periphyton initially and again after the periphyton mat (biofilm) is well established (Fernandes and Esteves 2003). The August 2012 standing crop of heterotrophic bacteria counts on natural LCR substrates were typical compared to other large North American rivers at 1.5 - > 2 x10^6 CFU/cm^2. Similarly, the fungal counts were in the typical to productive range on natural substrates (Table 3-8). Anaerobic sediments beneath the cobble armour at S4 gave moderate HTPC and fungal counts. More samples would be required to characterize the biofilm from erosional and depositional LCR sediments.

Table 3-8: Biofilm Component Samples from Natural LCR Substrates, Summer 2012)

Lower Columbia River		Shallow Natural Substrates		
18-Aug-12	units	S2 sand/cobble	S5 cobble	S4/S7 anaerobic
Heterotrophic Plate Count	CFU/cm^2	> 2000000	> 2000000	1500000
Mould	CFU/cm^2	19000	100	3400
Yeast	CFU/cm^2	3000	<100	900

CFU = colony forming unit

3.3.2 Composition, Abundance and Biomass of Periphytic Algae

Our sampling effort remained focused on the permanently wetted, shallow substrates in 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. Like most large rivers, LCR periphyton was dominated by diatoms representing between 61 and 97% of the average biovolume in all sample sites (Table 3-9). Over the years of study, the largest shifts in community structure occurred in the soft-bodied algae. For example, the flagellate group ranged from 1,160 cells/ml in 2009 to 172,000 cells/ml in 2008 (Scofield et al. 2011). Cyanobacteria were periodically prevalent. Filamentous cyanobacteria ranged from 1 - 40% by abundance, but that translates to only 0.01 – 3.2% of the total biovolume because of their small cell size.

Table 3-9: Range of Periphyton Relative Abundance and Biovolume Obtained from Artificial Substrates by Season 2008 – 2012 and Winter 2013

LCR	Summer 2008 - 2012		Fall 2008 - 2012		Winter 2013	
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	38 - 97	81 - 95	71 - 97	91 - 94	61	90
Flagellates	0.1 - 48	0.1 - 3	2.0 - 22	0.1 - 4.5	4.4	0.6
Cyanobacteria	1.0 - 40	0.01 - 0.8	1.3 - 13	0.01 - 0.4	33	3.2
Green	2.0 - 9	4 - 18	0.0 - 4	4.1 - 7	1.5	6

Filamentous green algae are slower growing and occurred most often on the sides of cobbles where there is more protection from scour and shear. They did occur on the artificial substrates and accounted for 4 – 18% of biovolume in summer months.

The nuisance invasive diatom *Didymosphenia geminata* (Didymo) is naturalized in LCR. Didymo frustules were detected in the drift from the Kootenay and from all LCR sites except the furthest upstream WQIS1. It seems unlikely that Didymo was contributed by the ALR.

Table 3-10: LCR Productivity Metrics (biovolume cm3/m2 : chl-a µg/cm2±SD) by Year during Summer and Fall 2008 – 2012, and Winter 2013

Year	Summer	Fall	Winter
2008	low 0.84±0.76 : 0.35±0.75	high 6.66±9.5 : 9.39±7.6	
2009	moderate 6.13±4.73 : 1.00±0.8	high 8.42±7.1 : 3.38±2.45	
2010	high 8.9±14.1 : 6.16±4.71	very high 16.5±9.6 : 26.04±14.2	
2012	moderate-low 1.67±1.3 : 1.46±0.44	moderate-low 1.99±1.7 : 2.08±0.91	
2013			very high 13.26±5.82 : 8.15±4.55

Large variations in the periphytic species assemblage and production metrics were observed among the years of study (Table 3-10; Appendix A1 & A2). 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 Arrow Lakes Reservoir. A total of 60 taxa were frequently observed in LCR studies. When all growth metrics are considered, the lowest periphyton growth year was 2012 (record high flows), and the highest year was 2010 (typical flows; Table 3-12). When the seasons are considered individually, the lowest measured growth occurred in summer 2008 and the highest occurred in fall 2010.

Species diversity and the Simpson's index indicate that LCR biodiversity is stable and moderate compared to other rivers (Appendices A1 & A2). However, 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, permanently submerged samplers (Table 3-13). It had more light and lower velocities (less shear) than the deeper sampler positions. However, many of the deep samplers also had high diversity but with increased numbers of tightly attached or stalked diatoms. (Appendices A1 & A2). Species richness was lowest in the fall at the deep sites and highest in the summer at the shallow sites. Average species richness at a single site ranged from 27 ± 10 to 43 ± 9 in LCR samples, with an overall average of 35 ± 4 taxa. The diversity in LCR is far higher than the diversity observed in the MCR, suggesting that the LCR has greater stability.

LCR production metrics for biovolume and chl-a were correlated, as expected (e.g., R=0.80 for averages presented in Table 3-13). Reported abundance and biovolume and chl-a consider live periphyton. Only AFDW includes live and dead material.

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. The summer period of flood years such as 2011 and 2012 included very high flows that apparently limited periphyton production. Summer 2012 production was distinct from summer 2008 – 2010. The highest biovolume and diversity occurred on the shallow samplers (always submerged) and declined with increasing depth (Figure 3-15).

Across all years, fall production was higher than the summer and declined with depth, most likely through decreased available light and increased velocity (Table 3-11). The shallow sites also showed lower productivity and diversity, probably because of periodic dewatering during the fall fluctuating flows. The fall 2012 periphyton metrics were similar to those from earlier years. Although production was high, species diversity was lower in fall than in summer 2012.

Winter production was measured in 2013 only and was very high, exceeding both the summer and fall of 2012. Mean periphyton production appeared to increase with depth, despite the lower available light in winter (Figure 3-15; Table 3-11). Stable winter flows benefitted Didymo growth and apparently benefitted other components of the periphyton community as well. The percentage of dead diatom frustules was also higher at 7.7% in the winter, compared to 4.0% in the fall and 3.4% in the summer. Winter 2013 conditions, including low stable flows, allowed thicker periphyton and allowed dead frustules to remain on the substrates.

Table 3-11: LCR Productivity Metrics (biovolume cm^3/m^2 : chl-a $\mu g/cm^2$ ± SD), by Sample Depth during Summer and Fall 2008 – 2012 and Winter 2013

Depth	Summer	Fall	Winter
shallow	moderate-high	very high	moderate-high
S	6.05±12.6 : 2.1±12.88	11.92±12.3 : 11.96±14.39	9.88±5.48 : 6.42±4.18
mod-shallow	low	moderate	very high
MS	1.34±0.67 : 1.37±0.55	3.20±1.47 : 2.11±0.74	13.38±6.21 : 8.00±5.33
moderate	moderate-high	high	very high
M	4.79±6.06 2.67±4.26	7.57±6.6 : 9.71±11.73	14.12±6.38 : 7.08±3.37
mod-deep	low	moderate-low	very high
MD	1.24±0.78 : 1.39±0.29	1.49±1.35 : 2.42±1.33	12.64±6.21 : 9.78±5.14
deep	moderate	high	very high
D	3.23 ±3.98 : 2.36±3.48	5.51±6.65 : 9.07±10.88	16.25±4.33 : 9.49±6.73

NOTE: The MS and MD depths in this table have 2012/13 values only and therefore show lower values than the adjacent depths that were sampled from 2008 – 2012/13.

Didymosphenia geminata or Didymo was particularly prevalent in 2013 as it had been in the late winter of other years (Dr. G. Martel, BC Hydro, pers. comm.). Its large mats flourished in the stable low winter flows and were dislodged after higher RBT flows commenced in April. Didymo attachment filaments were the dominant organic material in the winter samples and increased the ash-free dry weight (AFDW) dramatically. The overall summer and fall 2012 AFDW were similar at 0.34 ± 0.06 mg/cm^2 and 0.38 ± 0.17 mg/cm^2 respectively, while winter 2013 measured 2.77 ± 0.36 mg/cm^2 (Appendix A3). The highest winter AFDW measurements occurred at mid-depths and coincided with the thickest Didymo growth. Interestingly, the filaments were rarely colonized by periphytic diatoms as they were in the MCR. Based on field observation, very little periphyton grew on substrates beneath a thick Didymo mat. As a result, chl-a, autotrophic index, species richness and diversity appeared to drop when Didymo was prevalent in the winter, particularly on the natural substrates because they had thicker Didymo mats.

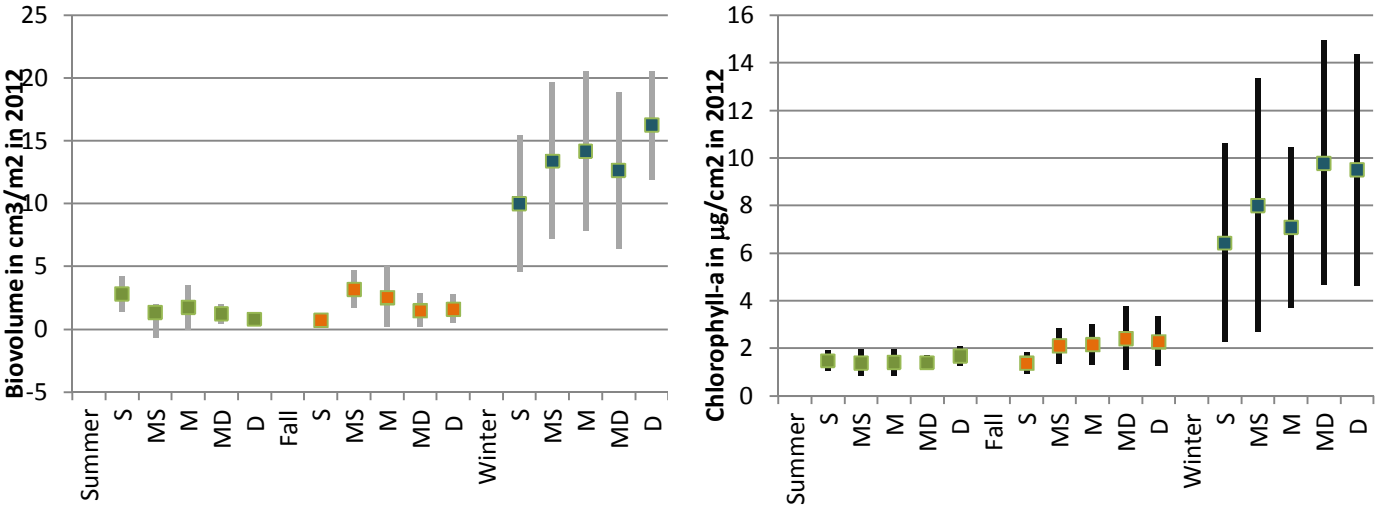

Figure 3-15: Mean Periphyton Biovolume (cm^3/m^2) and Chlorophyll-a (µg/cm^2) ± SD in Summer and Fall 2012, and Winter 2013 over the Range of Sampled Depths. Depth labels are: S=shallow, MS=moderately shallow, M=mid, MD=moderately deep, D=deep.

The winter season is clearly unique with increased 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.

Chlorophyll-a and AFDW provide complementary information that can be combined as a ratio into the autotrophic index (AI) (AI= AFDM (in mg/m^2) / chlorophyll a (mg/m^2) (Weber 1973). The autotrophic index is indicative of the proportions of the periphyton community composed of heterotrophic (fungi, yeasts, bacteria, protozoa) and autotrophic (photosynthetic bacteria and algae) organisms (Biggs and Close 1989; APHA 1995; Biggs and Kilroy 2000; Yamada and Nakamura 2002; Runion 2011). Higher values indicate a

greater proportion of non-photosynthetic organisms. In all seasons, greater autotrophic production occurred on the moderate deep and deep samplers, particularly at erosional sites (Appendix A3). These results imply that proportionately more photosynthetic production occurred at erosional sites, while more decomposition occurred at mixed and depositional site. 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/m^2) and AI values (2594 ± 920).

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. Filamentous green algae never occurred on substrates that were periodically exposed. Their growth was strongest in moderate-low flow periods and at the MS 1-2 m depths. Fall samples had the highest average autotrophic index values (Appendix A3).

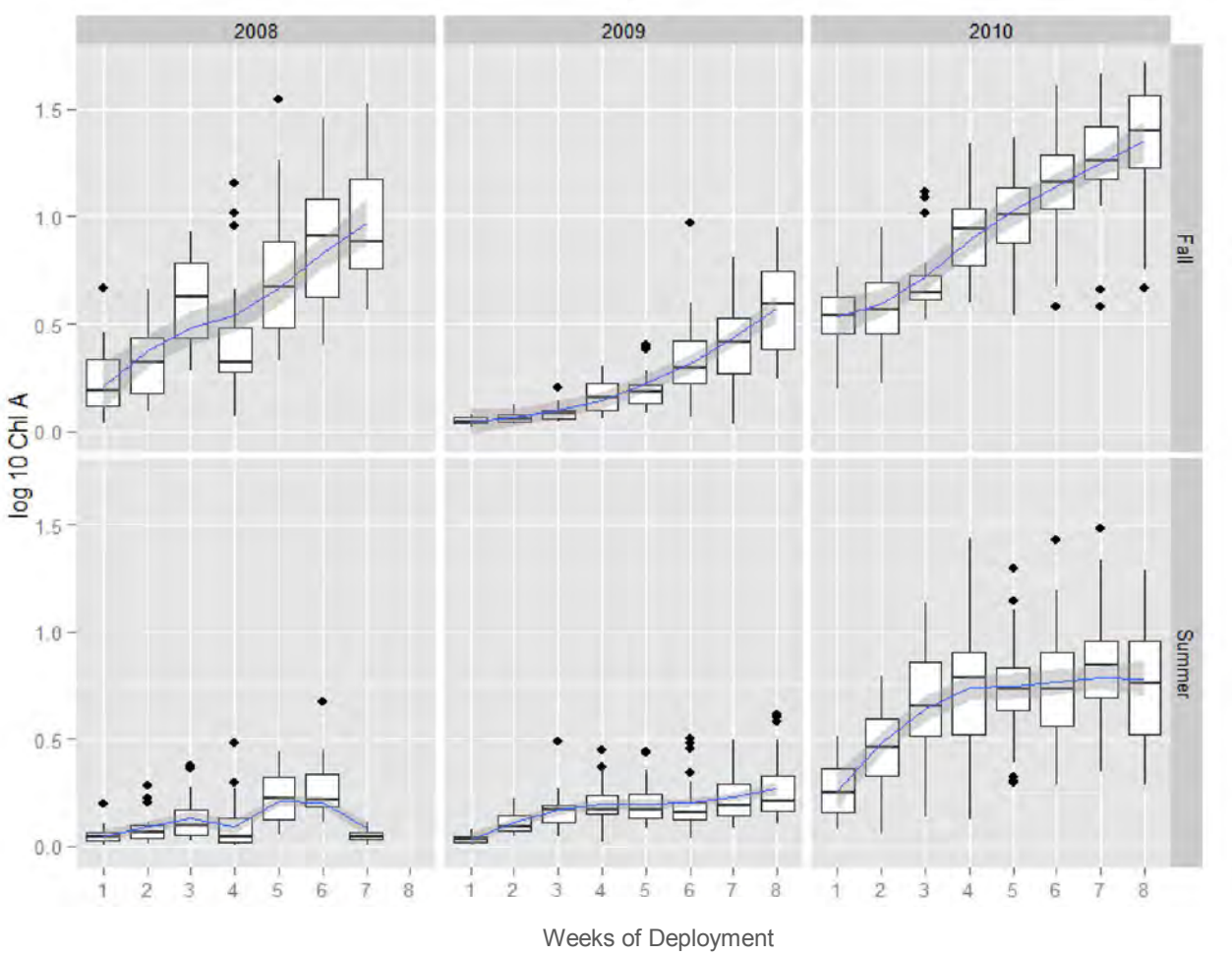

Figure 3-16: Weekly Periphyton Chl-a Accrual Rates (2008 – 2010) in the Summer and Fall. Fitted lines were generated using a locally weighted polynomial regression method (LOWESS). Data obtained from Scofield et al. 2011.

Scofield et al. (2011) completed time series research in 2008 – 2010. They concluded that the accrual time required for LCR periphyton to reach peak biomass on the closed cell Styrofoam was 6-7 weeks in the summer and more than 8 weeks during the fall fluctuating flows. The accrual chl-a data is graphed in Figure 3-16. In summer accruals, growth was leveling off by 4-5 weeks in all three years of study. By that time, periphyton losses and gains were matched, perhaps through flow-induced shear and through grazing. Benthic invertebrates were frequently observed on the samplers, particularly in the summer.

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.

Table 3-12: LCR Productivity Metrics for Winter 2013 Samplers Deployed for 12 Weeks and Long-term for 26 weeks

Upper varial zone,	artificial substrate LCR Styrofoam					
LCR	S1 MS winter	S1 M winter	S2 M fall	S1 MS winter	S1 M winter	S2 M fall
incubation time	12 weeks	12 weeks		26 weeks	26 weeks	26 weeks
abundance cells/cm^2 x 10^5	18.80	22.90	*not*	35.10	38.82	5.79
biovolume cm^3/m^2	14.70	23.60	*retrievable*	38.50	43.12	1.08
chlorophyll-a µg/cm^2	10.84	10.91		8.54	5.88	2.96
Number of samples	1	1	1	2	2	1

Long-term samplers were deployed to clarify the accrual curves, but several were not retrievable. In the 2013 winter deployment, we found greater growth on samplers deployed for 26 weeks than we did on samplers deployed for 12 weeks and retrieved at the same time (Apr 1, 2013). The 26 week samplers had roughly double the abundance and biovolume of the 12 week samplers (Table 3-12). The 26 week samplers also had far more bacteria that would presumably include decomposers and may have lowered chl-a. Chl-a would be further lowered by the Didymo cover, highlighting the need to use multiple production metrics to clarify trends. Although the number of retrieved 6 month samplers was low, it was clear that periphyton production continued to increase in the winter beyond the 12 week incubation period.

The periphyton data was also evaluated for the effect of the sample locations. Periphyton community structure and production were consistent among the erosional fast-flowing sites with cobble substrates (e.g. S1 S2), and were distinct from the depositional site S6 with its lower flows and silty substrates. Mixed sites had cobbles and fines. They were more erosional during high flows and more depositional during low flows. The erosional sites were dominated by rapid colonizing diatoms with firm attachment strategies. The depositional and mixed sites included more species, particularly of the motile diatoms (e.g. *Nitzschia*) that can re-position their cells as sediments deposit. Like the lab results, a microscope review of the depositional samples resulted in lower bacterial counts than in the erosional sites. Increasing heterotrophic dominance (i.e., sites dominated by decomposer microflora and non-viable organic materials such as dead cells and detritus) were more common in the depositional and mixed sites.

Didymo mats were rarely encountered on the depositional substrates. Site 6 was highly productive in the summer and fall seasons but not in the winter (Table 3-13). The shallow (S) and moderately shallow (MS) samplers were partially buried in sediments deposited during the winter deployment. Summer samples from Site 5 had low diatom counts in the shallow samples but were very high in bacteria and pico/nano flagellates, suggesting extensive decomposition at this site.

Table 3-13: LCR Productivity Metrics (biovolume cm^3/m^2 : chl-a $\mu g/cm^2$±SD) by Sample Site during Spring and Fall 2008 – 2012 and Winter 2013

Site	Type	Summer	Fall	Winter
S1	erosional	moderate 2.84±2.59 : 2.21±2.46	high 8.28±6.22 : 12.14±11.68	high 15.1±4.95 : 9.55±2.13
S2	erosional	high 7.16±15.5 : 2.66±4.32	high 9.96±7.02 : 8.39±8.92	moderate 14.54±6.08 : 3.62±1.78
S3	mixed	moderate 2.74±2.32 : 2.66±3.62	moderate 5.12±6.02 : 5.33±7.48	high 16.06±5.11 : 8.73±2.05
S4	mixed	moderate 3.36±4.27 : 1.68±1.91	high 9.06±10.27 : 7.70±7.72	high 12.6±5.32 : 11.95±3.54
S5	mixed	low 2.0±2.26 : 1.05±1.16	high 10.31±12.83 : 9.04±11.63	high 16.38±5.28 : 6.57±3.17
S6	depositional	high 6.84±7.11 : 4.03±5.8	high 5.17±7.01 : 12.76±17.11	low 5.57±4.19 : 4.15±2.37
S7	erosional	moderate 4.91±6.6 : 1.27±1.31	moderate 7.14±11.07 : 9.27±14.59	high 12.62±4.84 : 12.51±6.53

When all sites are considered, the summer deployment with 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 deployment would account for (Figure 3-17). Production between erosional and depositional sites was similar in the spring and fall but erosional sites had far more productivity in the winter than depositional sites.

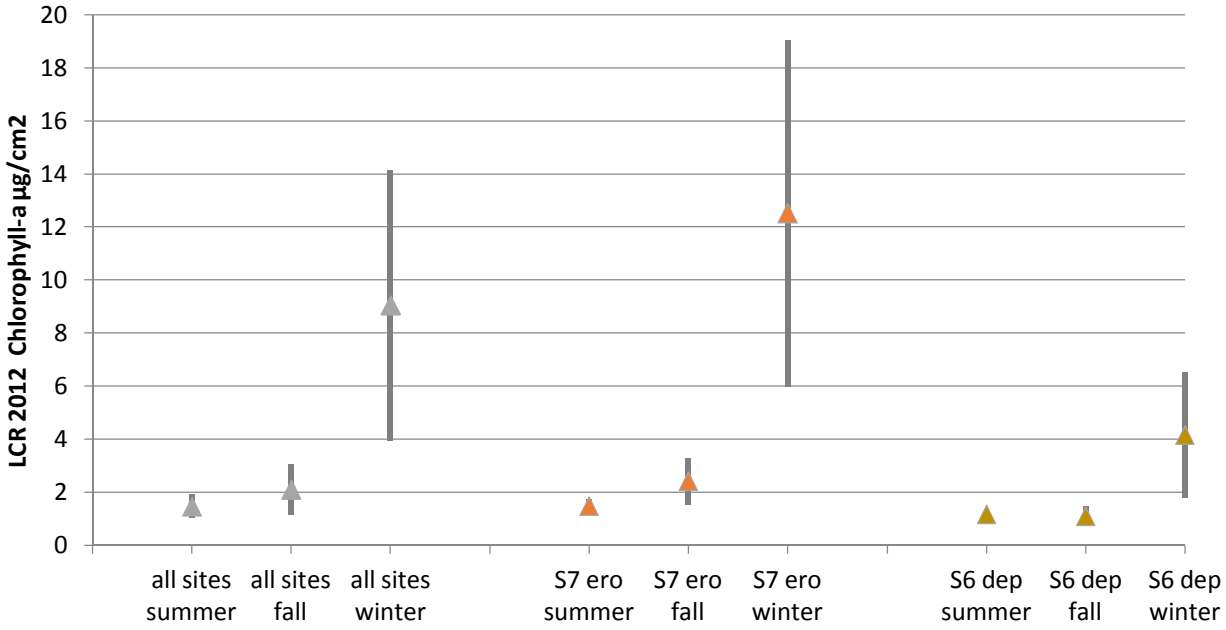

Figure 3-17: Average Chlorophyll-a (µg/cm2)±SD in LCR from All Sites, Erosional Site S7 and Depositional Site S6 during Summer 2012 (11 weeks), Fall 2012 (10 weeks) and winter 2013 (12 weeks)

Algae drifting in the LCR water column were dominated by cells coming from the reservoirs above the dams. Drifting algae settled onto the periphyton and remained viable, increasing periphyton diversity. Algae density in the drift during the May/June period was low in all samples, while the highest densities occurred in the September/October period (Figure 3-18). The fall drift composition was also more variable, with more soft-bodied green and flagellated algae. The fall periods were affected by deeper mixing of the reservoir water columns and by the expected arrival of ALR fertilizer nutrients (Schindler et al. 2006). Over 60% of the drift diatoms and filamentous cyanobacteria were exclusively lake forms and they composed approximately 1-10% of LCR periphyton after they became incorporated into the periphyton biofilm. For example, *Diatoma spp*. and *Tabellaria spp.* originated in the reservoirs and these genera are known to persist in the periphyton of downstream rivers that have stable flows (Bonnett et al. 2009).

Drift in the Kootenay River had more algae cells and chl-a than LCR drift and would benefit LCR periphyton production below the confluence. For example, the chl-a was 2.0 µg/L in the Kootenay but was only 0.4 µg/L in LCR above the Kootenay confluence on August 17, 2012. The impact of the more productive Kootenay was still detectable in LCR drift and periphyton at Genelle WQIS5. Kootenay taxa such as *Fragilaria crotonensis*, were common at Genelle, as were species originating from the ALR such as *Synedra acus* and *Synedra nana*. These are large diatoms and it is interesting that they could remain suspended over many kilometers of river flow. In every case, chl-a increased from the upstream WQIS1 to the downstream WQIS5 samples by an average of 65% in the summer and fall months.

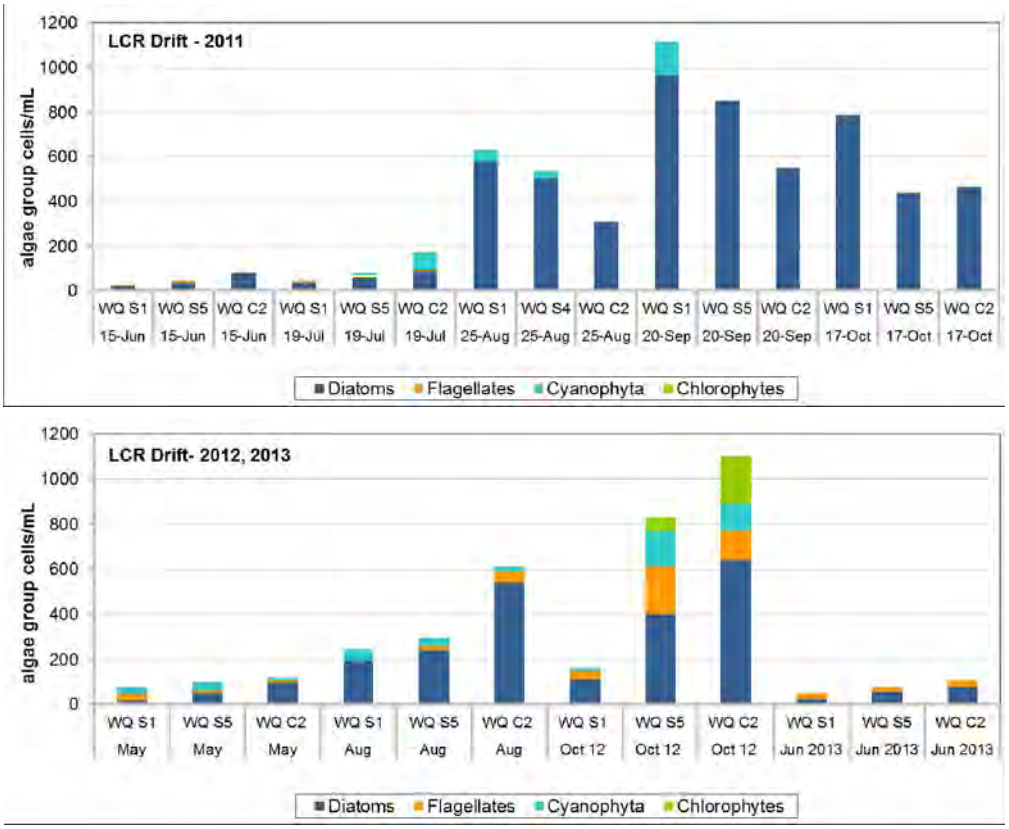

Figure 3-18: Composition of Drift Samples from LCR Collected Between 2011 and 2013

All plankton tow samples collected with an 80 micron mesh net included four large, dominant diatoms that were donated by both reservoirs, but in June 2013 amidst record flooding, the golden flagellate *Dinobryon* became prevalent in the plankton tows. The ALR donated *Dinobryon sertularia* at 10% of the plankton tows, while Kootenay Reservoir donated *Dinobryon bavaricum* at 46% and *D. bavaricum* still composed about 40% of the plankton tows from Genelle WQIS5. Flooding may have encouraged their growth since these algae can also feed on bacteria-sized particulates. *Dinobryon* was seldom detected

in the periphyton samples. The plankton hauls also contained numerous zooplankton donated to LCR by ALR, and large zooplankters can be utilized by fish.

3.3.3 Periphyton Community Groupings

Community analyses of the 2008 – 2013 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. Data were grouped by year (Year (ANOSIM, R: 0.65, p = 0.001), season (Season ANOSIM, R: 0.13 , p = 0.001), and possibly by depth (Transect ANOSIM, R: 0.02, p = 0.053) (Figure 3-19). NMDS analysis (stress = 0.17) of the five groups at the genus level indicated substantial overlap between sites (Year (ANOSIM, R: 0.01, p = 0.10).

The community analysis suggests that there were high inter-annual and seasonal variations. Over the sample period, no large-scale shifts in periphyton communities were observed and depth was a less important determinant of community structure than season or year. Although not explicitly tested, different taxonomists may account for some of the differences observed between years. The remaining differences observed are due to the general operating regime over the sample period. Winter communities are distinctly different than summer or fall communities, although only one year of winter data is currently available.

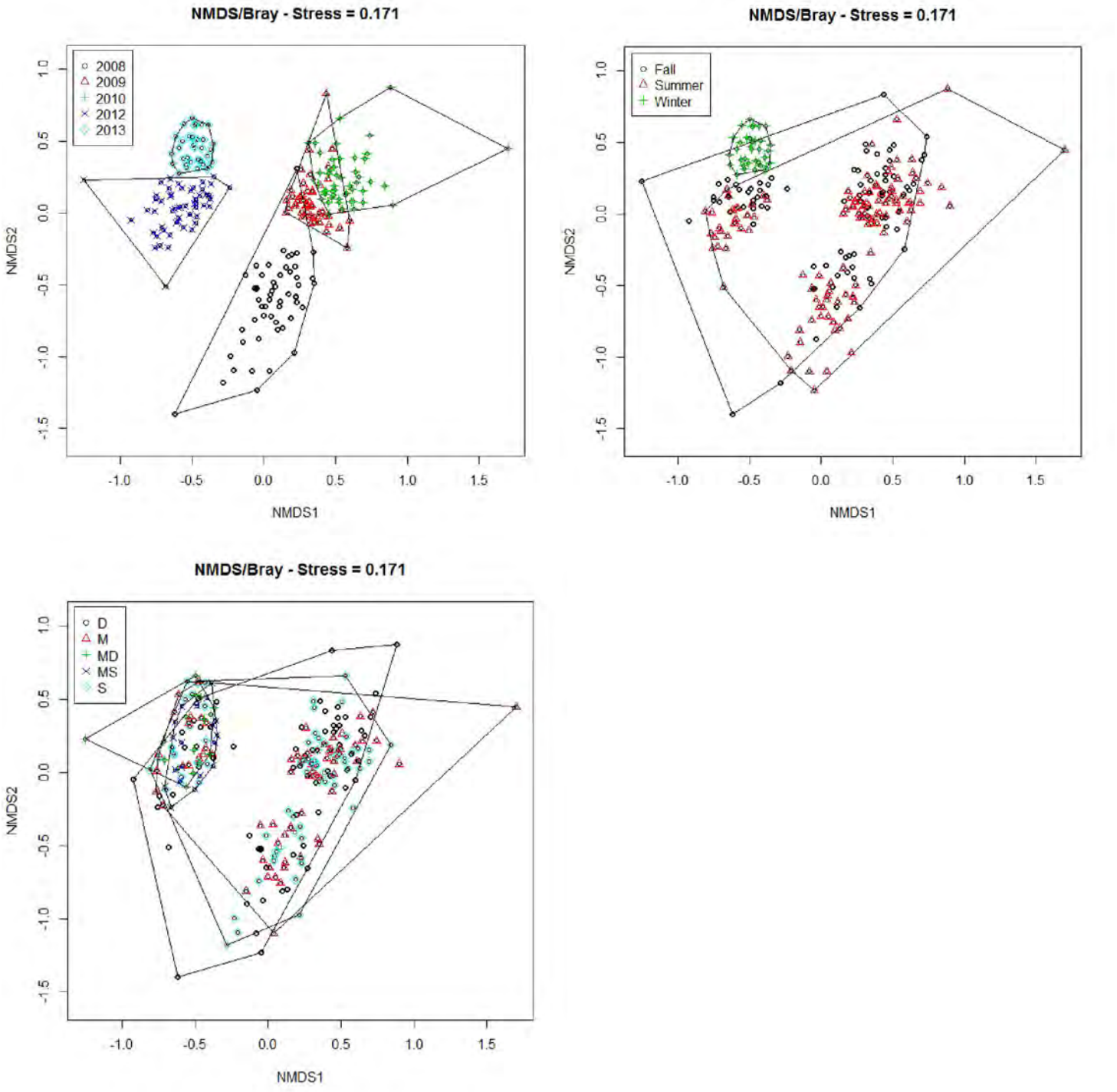

Figure 3-19: NMDS of Periphyton Abundance at the Genus Level Grouped by Year, Season, and Depth for all Data Collected between 2008 and 2013

3.3.4 Periphyton Production Models

Model averaging of key periphyton responses included abundance, biovolume, chl-a, and Simpson's Index and was only completed on Reach 2 data since this reach was sampled during all years. For each response, the following explanatory variables were used: mean daily maximum water temperature, site type (erosional, depositional, or mixed), substrate, velocity, mean daily maximum light intensity, HLK flow, the first principle component of an analysis of reservoir conditions that represented reservoir temperature and elevation, and depth (transect). There were numerous plausible models (those with an AICc<3.0) for abundance. There were 9 and 12 models for abundance in the summer and fall respectively. Similarly, there were 18 and 8 biovolume, 11 and 12 chlorophyl, and 18 and 12 Simpson's index models with a ΔAICc<3 in the spring and fall respectively. The proportion of variance explained by models was generally good for periphyton responses. THe R^2 for abundance models was R^2 ~ 0.28 in the summer and 0.46 in the fall. Similarly, the biovolume was R^2 ~ 0.30 the summer and 0.58 in the fall. Chl-a variance explained was R^2 ~ 0.75 in the summer and 0.62 .fall. FInally, Simpson's Index variance explained was R^2 ~ 0.39 in the summer and 0.22 in the fall (Figure 3-20). For each response, the variation described was roughly the same for the best model during the summer and fall periods.

During the summer months, several key trends were observed. The effects of mean daily water temperature was the most important predictor of periphyton abundance, biovolume, and chl-a. Interestingly, measures of periphyton production (abundance, biovolume, chl-a), were all negatively correlated with temperature (Figure 3-20).

Other key parameters predicting periphyton production included site type, and substrate, which were also negatively correlated meaning that productivity decreased as sites became more erosional in nature and as substrate size increased. The relative strength of site type and substrate varied with response. For biovolume models, site and substrates had the greatest effect size and relative variable importance when compared to other measures of production. Simpson's index was different than other measures of production, with site type and reservoir conditions being the most important determinants of periphyton species diversity. Generally, periphyton diversity was greatest at moderate to shallow sites and decreased as sites became more erosional in nature. Although velocity was not an important predictor of periphyton productivity in the summer, a trend emerged of decreasing productivity with increasing velocity but this trend was dependent upon other conditions in the river such as depth and substrates.

During the fall fluctuating flows, substrates, depth, and light appeared to be the most important determinants of periphyton productivity. Generally, productivity increased with increasing substrate size, decreasing depths, and increasing light intensity. This infers that the most productive zones on the river occur along stable, cobble river banks at moderate to shallow depths. Further, since the importance of light to periphyton production increases during the fall, light availability may act as a limiting factor on periphyton growth during this time. This is supported by the longer time required to reach peak biomass (Figure 3-16 and Scofield et al. 2011).

Periphyton diversity appears to be greatest in erosional areas with cobble substrates during the fall. Interestingly, this trend is the opposite of the summer, when the greatest diversity was more apparent at depositional sites. Finally, reservoir condition is also an important determinant of periphyton diversity, although this has not been fully explored, but may relate to nutrient and algae species donations in reservoir flows.

Discharge from HLK appeared to have a highly variable effect on periphyton production, although it did not appear in many of the top predictive models. The wide variations in this explanatory variable limit its predictive value for periphyton productivity. Future iterations of the model will attempt to explain potential interactive effects between HLK and BRD flows (not currently included in the model), effects of ramping, and daily variability on periphyton production.

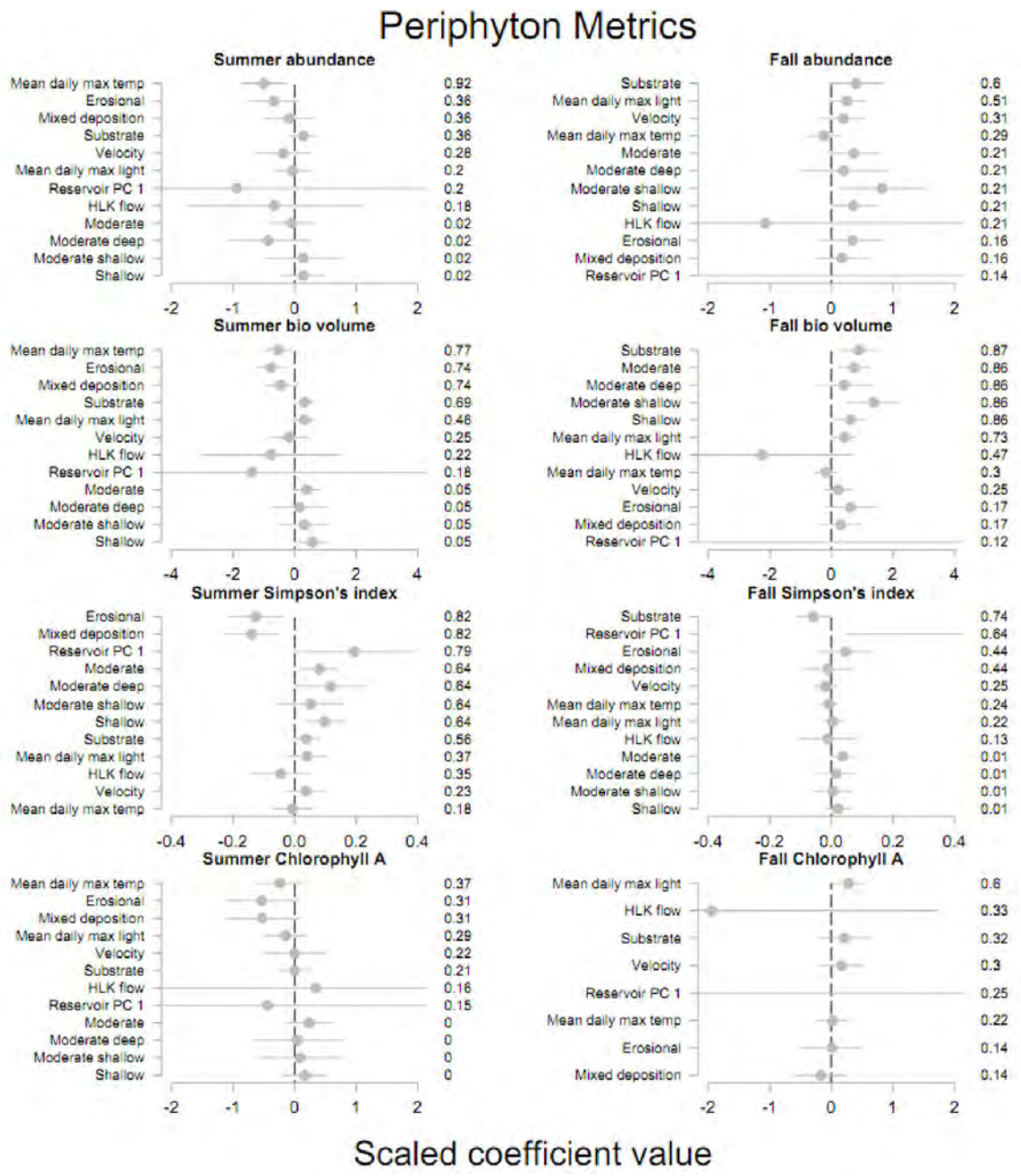

Figure 3-20: Mean coefficients and their 95% confidence limits of standardized explanatory variables of periphyton production in LCR during the summer and fall. Periphyton responses included abundance, biovolume, chlorophyll A, and Simpson's index. Explanatory variables included Mean Daily Maximum Water Temperature, Site Type (Erosional, Depositional, or Mixed), Substrate, Velocity, Mean Daily Maximum Light Intensity, HLK Flow, the first principal component of reservoir conditions (reservoir temperature and elevation), and depth transect. 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 of each figure.

3.3.5 Comparison of Artificial and Natural Substrates

Quantitative natural substrates were collected from upper varial cobble substrates during 2012 and 2013 in an attempt to understand how well LCR closed cell Styrofoam substrate predicts growth on natural substrates. Periphyton from natural substrates usually has more variation than adjacent artificial substrates (Biggs 1996; Wetzel, 2001). Sand and cobble substrates had similar species lists to the artificial substrate samplers, with small changes in the proportions of the algae types. The natural cobble samples had higher proportions of slower-growing filamentous green algae and of filamentous cyanobacteria than the comparable artificial substrate samplers. The filamentous types were thickest on the sides of natural cobbles, presumably because there was less scour and shear in the interstices.

Growth metrics from the natural substrate samples are contrasted with the comparable Styrofoam artificial substrate samplers in Figure 3-21. When cobble substrates are compared to adjacent artificial samplers, the exaggeration of growth metrics on the artificial substrates ranged from no difference to 3 times greater. The typical sample pair showed twice the growth metric on the closed cell Styrofoam as on the adjacent natural substrate.

From observation, grazing was an important factor in LCR samples and may have been greater on natural substrates than on Styrofoam substrate. Further, the closed cell Styrofoam may provide easier attachment sites than natural substrates (Figure 3-22). Interestingly, the natural substrates had about half the percent dead diatom count of the comparable Styrofoam substrates (2.6% vs. 3.7% in summer; 2.7% vs. 7.7% in winter. The number of pairs of natural substrate/Styrofoam samples successfully retrieved only supports limited statistical comparison.

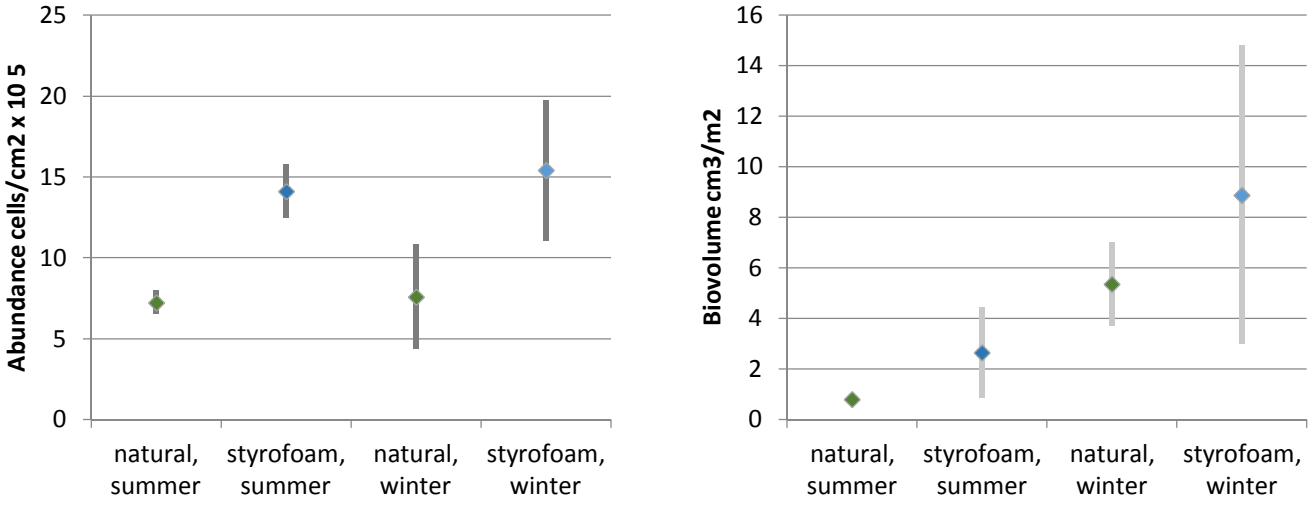

Figure 3-21: Average Periphyton Abundance and Biovolume from Natural and LCR Styrofoam Artificial Substrates Deployed for 10 -12 weeks in the Lower Columbia River in 2012 – 2013

Based on this limited research, it would appear that a correction factor of approximately 2 is required to directly compare the artificial sampler results to the natural substrates of LCR.

The long-term 6 month deployments of the artificial substrates are arguably closer to the incubation time of the natural substrates, but the production metrics were even further apart, however, the variations (deviations from the means) between replicate samples were very close, with only 6.5% more variation between replicate natural substrate samples than between replicate 6 month samples collected from closed cell Styrofoam.

Table 3-14: Periphyton Abundance on Open and Closed-Cell Styrofoam Deployed in Parallel in the MCR and LCR at Mid-transect Depths

Upper varial zone, 2012 Abundance cells/cm^{2} x 10^{5}	artificial Styrofoam substrate in MCR				artificial Styrofoam substrate in LCR					
	R4 BR T2	R4 BR T2	R3 S3 T2	R3 S3 T2	R2 S4 MD	R2 S4 MD	R2 S6 MD	R2 S6 MD	R2 S7 MD	R2 S7 MD
Styrofoam type	open	closed	open	closed	open	closed	open	closed	open	closed
Diatoms	4.59	0.75	1.76	0.25	3.55	2.67	1.76	1.68	5.13	3.57
Green	0.22	0.00	0.00	0.00	0.34	0.95	0.58	0.00	0.16	0.61
Cyanobacteria	6.84	1.28	4.80	3.45	8.17	5.21	3.03	3.58	5.47	6.15
Flagellates	1.96	0.83	1.25	0.82	2.21	1.54	1.21	0.88	1.58	1.24

The natural LCR substrate samples collected in the winter had a canopy of Didymo filaments with very little periphyton or other microflora attached to them. These samples also had fungal spores probably of the *Hyphomycetes.* Both the Didymo and the hyphomycetes increased the AFDW component and lowered the autotrophic index substantially on these natural substrate samples (Appendix A3). Didymo was also prevalent on LCR Styrofoam while fungi were not detected.

The use of Styrofoam as an artificial substrate is less common than unglazed or stone tile (Cattane and Amireault 1992, Biggs and Close 1989, Biggs and Kilroy 2000). The differences between the closed-cell MCR Styrofoam and the open-cell LCR Styrofoam, and the performance of stone tile as an alternate artificial substrate were also investigated in mid-depth LCR and MCR samplers during 2012 (Figure 3-22). In side-by-side deployments, the open-cell MCR Styrofoam always had more periphyton than the closed-cell in all river environments (Table 3-14). The difference between the two styrofoams was greater in the stressed slow-growing MCR than in the moderately productive LCR. For example, in the MCR deployments, the average difference between the open and closed-cell Styrofoam was 84% for diatoms, 100% for greens, 60% for cyanobacteria and 49% for flagellates. However, in LCR deployments the difference between the two styrofoams was only 21% for diatoms -44% for greens, 10% for cyanobacteria and 27% for flagellates. We presume that the rough surface of the open cell Styrofoam provides more attachment sites/refuge and its roughness snags more drifting algae than the closed-cell Styrofoam.

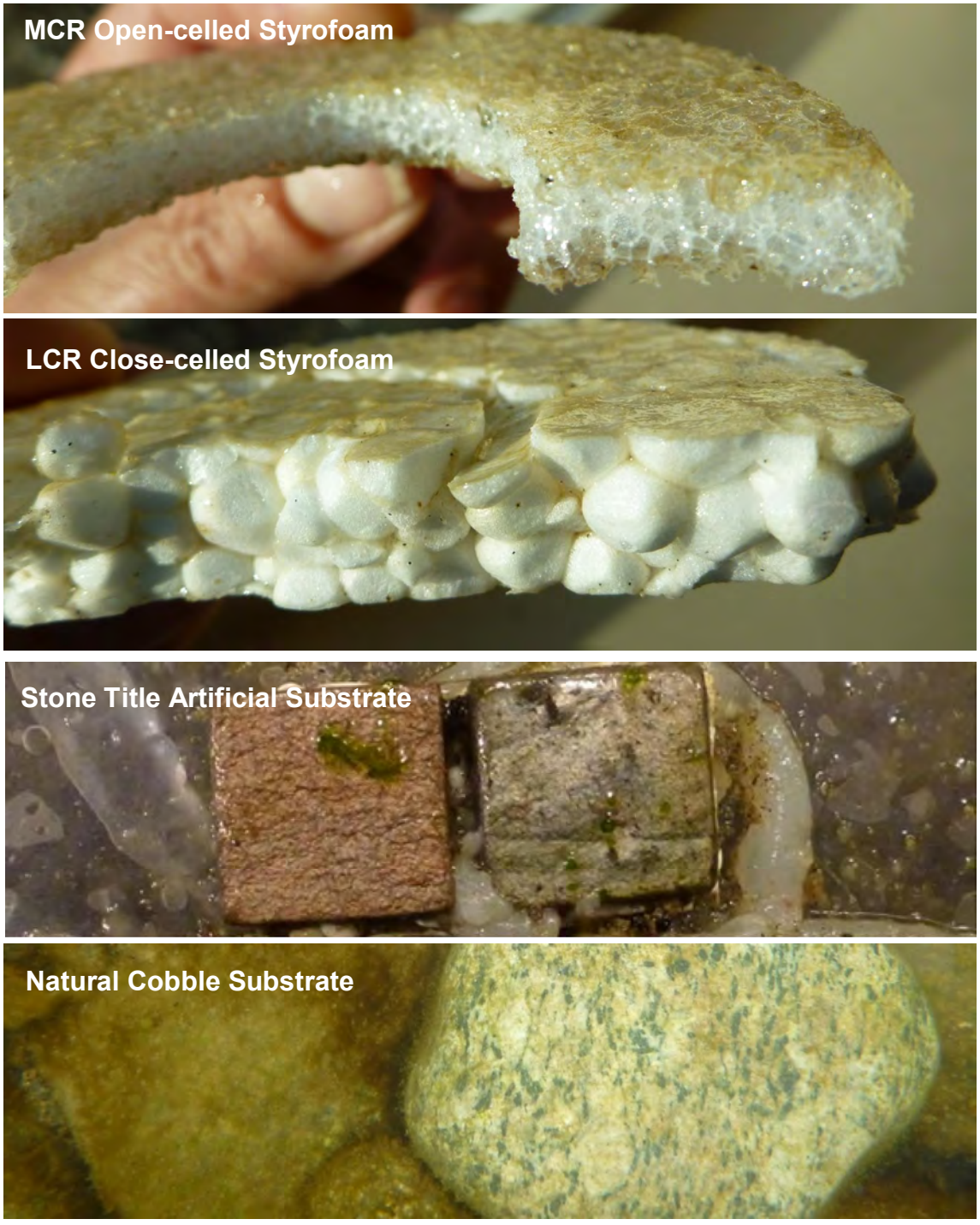

Figure 3-22: Comparison of artificial and natural substrates after deployment for 10 - 12 weeks in the Lower Columbia River in 2012 – 2013

Unglazed or stone tile is widely used as an artificial substrate in riverine periphyton studies (Biggs, 2000). In LCR, stone tile proved to be a viable alternate substrate in both the field and the lab. Variations in the periphyton communities seen on the tile were less than those on the natural substrate, likely through fewer microhabitat variations. The closed-cell Styrofoam and the tile substrates gave results that were very close in terms of both abundance and biovolume. The average abundance of eight fall 2012 parallel samples was 10.6 ± 2.8 cells/cm^2 x 10^5 for closed cell Styrofoam and 10.1 ± 4.6 cells/cm^2 x 10^5 for tile. Similarly, the average biovolume of eight fall 2012 parallel samples was 2.28 ± 2.12 $10^3/m^2$ for closed cell Styrofoam and 2.39 ± 1.48 $10^3/m^2$ for tile (Table 3-15). There were no significant differences for abundance (t = -0.22, p = 0.83) or biovolume (t = 0.11, p = 0.91). Like natural cobble substrates, the range in abundance was greater in the tile samples. These data suggest that results from LCR closed cell Styrofoam artificial substrate should be comparable to results obtained in the literature from tile artificial substrates.

Table 3-15: Periphyton Growth Metrics on Stone Tile and Closed-Cell Styrofoam Artificial Substrates Deployed in LCR for 10 Weeks in Fall 2012

Upper varial zone, Fall 2012	artificial substrate - stone tile							
LCR	S1 M	S3 MS	S3 M	S4 M	S4 MD	S5 M	S6 M	S7 M
incubation time 10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks
abundance cells/cm^2 x 10^5	13.1	9.7	14.1	14.8	9.51	0.24	11.4	8.22
biovolume cm^3/m^2	2.60	5.04	1.77	3.37	2.43	0.34	2.78	0.77
chlorophyll-a µg/cm^2	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
number of samples	2	2	2	2	2	2	2	3

Upper varial zone, Fall 2012	artificial substrate - closed-cell LCR Styrofoam							
LCR	S1 M	S3 MS	S3 M	S4 M	S4 MD	S5 M	S6 M	S7 M
incubation time 10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks	10 weeks
abundance cells/cm^2 x 10^5	13.5	8.9	10.3	7.9	9.7	7.6	10.7	15.8
biovolume cm^3/m^2	7.36	1.56	2.24	1.04	1.83	0.73	1.21	2.30
chlorophyll-a µg/cm^2	2.53	1.24	3.29	1.23	1.45	2.2	1.09	2.64
number of samples	1	1	1	1	1	1	1	1

3.3.6 Periphyton Support of Higher Food Chain Components

The periphyton and the drift are extensively used by benthic invertebrates in LCR. The effects of grazing and the grazers themselves were frequently observed on artificial and natural substrates (Figure 3-23). We observed a marked drop in benthic invertebrate density on cobbles coated with Didymo compared to adjacent substrates that were clean.

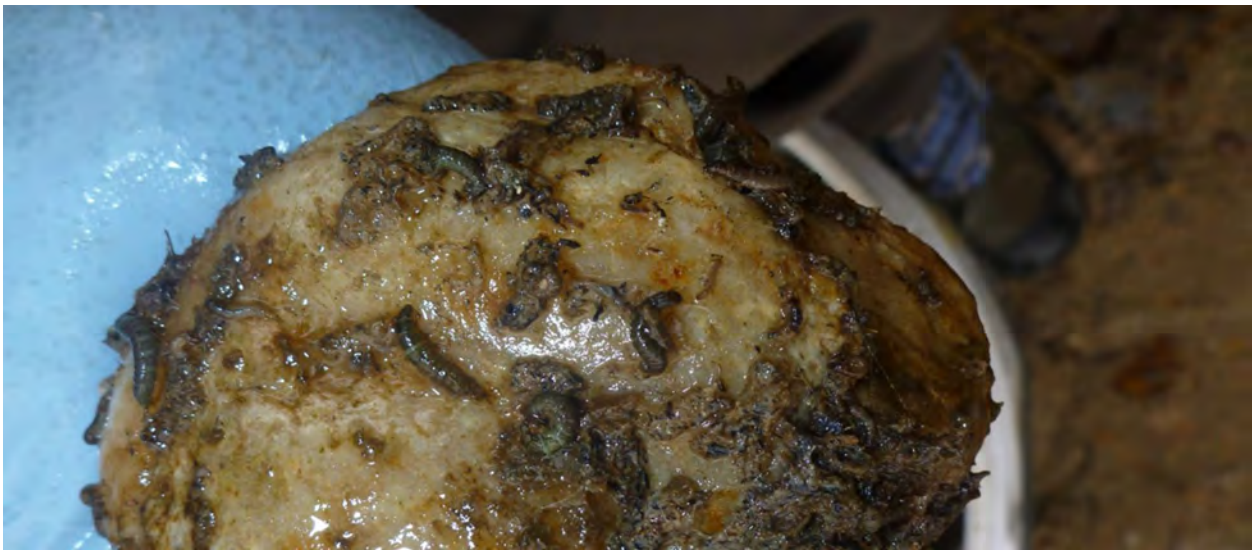

Figure 3-23: Benthic invertebrates grazing on periphyton on a cobble from LCR

The 60 common periphyton taxa identified in LCR were ranked by their edibility to benthic invertebrates. The dominant periphyton species by biovolume are listed in Table 3-16.

LCR drift contained large or colonial diatoms originating from LCR or Kootenay Lake. These taxa settled onto the periphyton. They had fair or poor forage quality because their diatom cells or colonies were too large for many of the smaller filter feeders and benthic invertebrates to utilize. However, the drift also contained bacteria and detritus that is a food supply for smaller invertebrates. The forage quality of the in situ periphyton taxa ranged from good to poor. The large filamentous green periphyton species such as *Ulothrix*, and *Cladophora* may not be directly edible, but they can harbour important food organisms. Unlike Didymo filaments, green filaments act as “ecosystem engineers”, providing habitat opportunities for smaller organisms in LCR (Konrad et al. 2011).

Table 3-16: Dominant LCR Periphyton Taxa, their Source and their Forage Quality for Benthic Invertebrates, 2008 - 2013 data

Type	Dominant Taxa	Habitat	Main source	Forage quality
diatom	*Achnanthidium spp.*	periphyton	LCR	good
diatom	*Tabellaria spp.*	plankton and periphyton	KL	fair
diatom	*Synedra ulna, nana (lrg species)*	phytoplankton	ALR	fair
diatom	*Synedra ulna (small varieties)*	periphyton	LCR	good
diatom	*Fragilaria crotonensis*	phytoplankton	KL ALR	poor
diatom	*Fragilariforma.F. intermedia*	plankton and periphyton	KL	fair
diatom	*Diatoma elongatum*	plankton and periphyton	LCR ALR	fair
diatom	*Frustulia spp.*	periphyton	LCR	fair
diatom	*Didymospenia geminata*	periphyton	LCR KR	poor
diatom	*Eucocconeis flexella*	periphyton	LCR	fair
diatom	*Cocconeis spp.*	periphyton	LCR	fair
green	*Ulothrix spp.*	periphyton	LCR	fair - lrg
green	*Eremosphaeria spp.*	periphyton	LCR	poor
green	*Cosmarium spp.*	periphyton	LCR	poor

LCR = Lower Columbia River KL = Kootenay Lake ALR = Arrow Lakes Reservoir KR = Kootenay River

3.4 Benthic Macroinvertebrates

3.4.1 Macroinvertebrate Rock Basket Recovery

During the three sampling sessions in 2012/13, 90% of rock baskets were recovered. As with the periphyton substrates, the greatest losses occurred during the summer sampling season when flows were unusually high. The high flows caused the movement of logs and debris that caused lost samplers or entanglements that prevent them from being dislodged from the river bottom.

Because summer retrieval coincided with fall deployment, there were fewer sampler apparatuses available for fall deployment. Site 2 (S2) was the most problematic; with three summer samplers entangled. Consequently, samplers were not deployed at S2 during the fall session. Conversely, the lower, steady winter flows facilitated a 100% recovery (Table 3-17).

Table 3-17: **Rock basket recovery by season in 2012/2013. Fractions indicate the number of substrates recovered over the number of substrates deployed.**

Season	2012/2013
Summer	27/35
Fall	28/30
Winter	35/35

3.4.2 Summary of 2012/13 Benthic Invertebrate Sampling

LCR had an abundant and diverse community of benthic macroinvertebrates. Rock basket sampling resulted in the collection of 42, 33 and 29 different taxa during the summer and fall of 2012, and winter of 2013, respectively (Appendix A4; Tables A6-7 through A6-9). 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 these larger groups.

The winter of 2013 was the first year that productivity data was collected to better coincide with MWF flows. The winter data was generally comparable to summer and fall. The highest mean abundance (# / basket) occurred in the fall with 7550 organisms per basket, followed by summer and winter with 4507 and 3980, respectively. However, the number of organisms per basket within each season was highly variable; the SD exceeded the mean for all three sampling seasons. Biomass (g/basket) data had a similar trend, and was also highly variable (Appendix A4; Tables A6-7 through A6-9).

Mean species richness numbers were very similar across the three seasons ranging from 25±9.3 in the summer, 21.73 ± 5.7 in the fall and 18.36 ± 4.6 in the winter (Appendix A4; Tables A6-7 through A6-9). Dominant taxa in the summer and fall included Hydropsychidae (net-spinning caddisflies), *Tvetenia discoloripes* gr. (chironomid) and *Brachycentrus occidentalis* (Mother's Day caddis). In the winter a different suite of organisms dominated, with *Simulium* (black flies) having the greatest abundance, followed by *Synorthocladius*, and *Orthocladius* Complex (both chironomids). 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 50.7 and 54.5 percent of the relative biomass. In contrast, Gastropoda (25.8%), Diptera (23.4%) and Trichoptera (23%) maintained the greatest relative biomass in the winter. This trend was also apparent when comparing EPT Richness and Percent EPT. The richness numbers for EPT were comparable across seasons, but the Percent EPT occurring in the winter was lower compared to summer and fall (14 compared to 59 and 34) (Appendix A4; Tables A6-1 through A6-3 and Tables A6-7 through A6-9).

The mean Hilsenhoff Biotic Index (HBI) was less than 5 for all seasons, indicating a greater predominance of pollution sensitive species (e.g., EPT). 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 the winter, there appeared to be a modest trend of increasing abundance of benthic invertebrates with increasing depth. This trend was not apparent in the summer or fall datasets that extended across multiple years (Appendix A4; Tables A6-4 through A6—6).

3.4.3 Yearly Comparisons of Benthic Invertebrate Sampling

Benthic invertebrate 2012 abundance and biomass data from the summer and fall sampling sessions were comparable to data collected in earlier years of the study. Generally, the whole dataset was highly variable and often maintained SD that exceeded the mean value (Appendix A4; Tables A6-1 through A6—9). Trends in depth, season, site or year were not readily apparent when just looking at the summary statistics.

Species richness tended to be the highest in the summer with a mean species richness of 24.5 averaged across the four years. This compared to 22.9 in the fall. Similarly, percent EPT was also higher in the summer, at 42% across the four years. In the fall, the prevalence of EPT was highly consistent across years, and encompassed approximately 33% of the organisms. On average, there were just over 5 and 6 different EPT taxa documented in the summer and fall, respectively. The occurrence of chironomids was more variable across years, but on average this group made up 29 and 45% of the summer and fall samples, respectively. Chironomids tended to be more abundant during the single winter sampling session (48%), especially when compared to summer. Together, EPT and chironomids encompassed more than 70% of the samples in the summer and fall, and approximately 62% in the winter.

Interestingly, the Simpson's Index tended to be considerably lower in 2012 during both the summer and fall sampling sessions compared to previous years, indicating a lower level of diversity in 2012 that may be indivative of the higher observed flows. The Hilsenhoff Biotic Index also tended to be lower in 2012.

3.3.4 Benthic Community Groupings

Community analyses of the 2008 – 2012 data were completed at the species level. Data were grouped by year (Year (ANOSIM, R: 0.19, p = 0.001) and season (Season ANOSIM, R: 0.16, p = 0.001) (Figure 3-24). Sites did not seem to group by depth (Transect ANOSIM, R: 0.008, p = 0.29)). Benthic communities in 2012 and 2013 similar to each other, but were different than previous years. These differences are potentially indicative of higher than average flows, and the inclusion of winter data for the first time in 2012 and 2013. Further, data from 2012 and 2013 contained additional sampling of two new depth sites which may also partially explains the difference in community observed between years.

The benthic community analysis suggests that there were high inter-annual and seasonal variations. Large-scale shifts in benthic invertebrate communities were not observed over the sample period. Differences in taxonomist may account for some of the difference between years but part of the annual variation is likely associated with the general operating regime over the sample period.

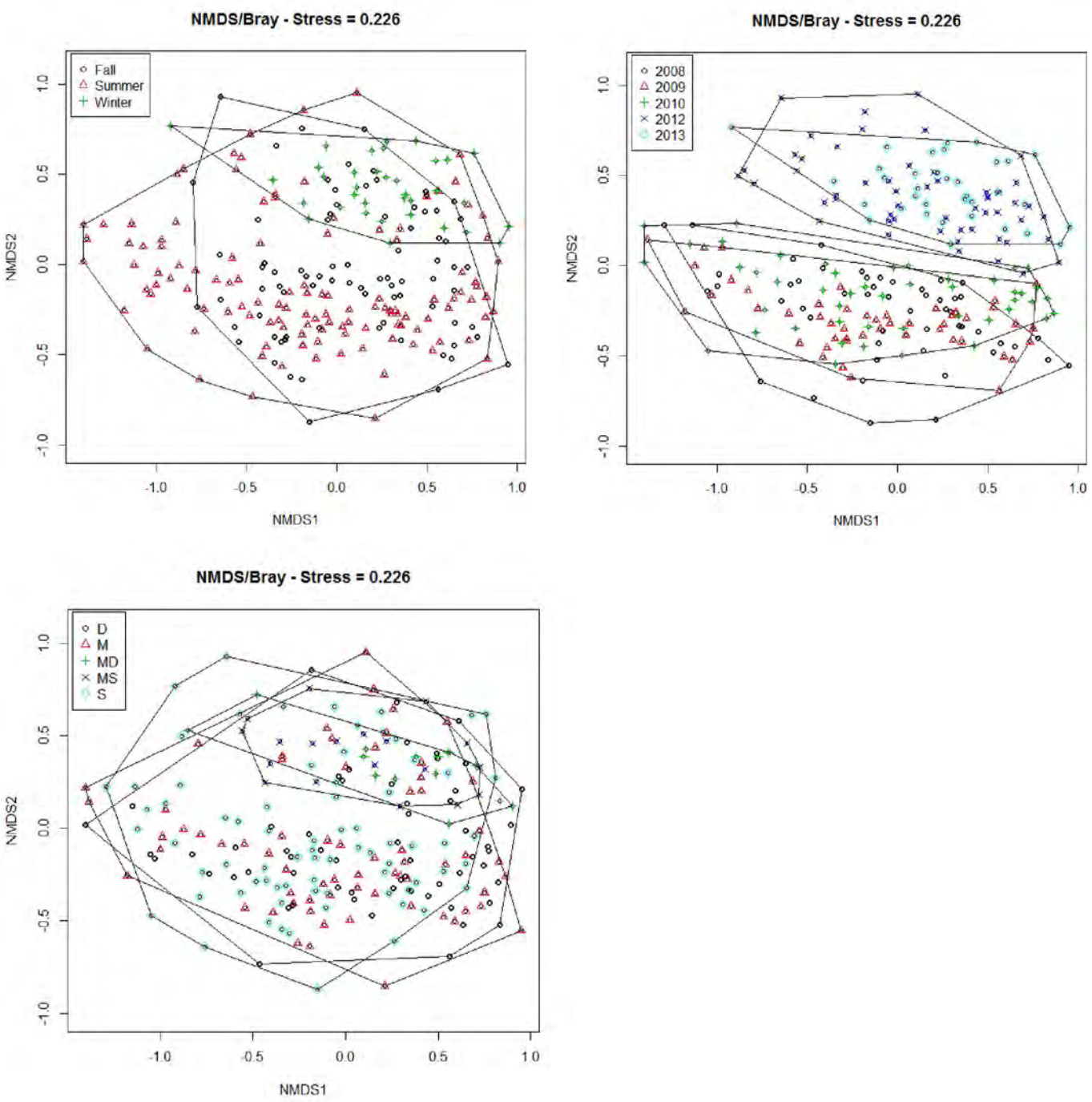

Figure 3-24: NMDS of benthic invertebrate abundance at the species level grouped by season, year and depth for data collected between 2008 and 2013

3.3.5 Benthic Production Models

Model averaging of key responses including abundance, biomass, Simpson's Index and Hilsenhoff's Index, was only completed on Reach 2 data since this reach was sampled during all years. For each response, the following explanatory variables were used:

Mean Daily Maximum Water Temperature, Site Type (Erosional, Depositional, or Mixed), Substrate, Velocity, Mean Daily Maximum Light Intensity, HLK Flow, the first principle component of reservoir condition representing an elevation and temperature gradient, and depth (transect) were used in models. Modeling of each response for spring and summer data suggested that there were numerous plausible models (those with an AICc<3.0). There was 11 models for abundance in the summer and 18 in the fall. Similarly, there was 13 and 19 top models for biomass, 7 and 8 models for Simpson's Index, and 11 and 18 models for Hilsenhoff Index in the summer and fall respectively. The proportion of variance explained varied by response, with Simpson's Index (R^2 ~ 0.60 (summer)- 0.85 (fall)) and Hilsenhoff Index (R^2 = 0.48 in summer and fall) having the greatest proportioned explained, while biomass (R^2 ~ 0.13 (summer)- 0.14 (fall)) and abundance (R^2 ~ 0.14 (summer)- 0.30 (fall)) had the least (Figure 3-25). The lower variances described for benthic abundance and biomass reduce our ability to understand relationships between river conditions and these measures of productivity.

The effects of season on benthic invertebrate communities are more difficult to interpret than for periphyton. The larger confidence limits on modelled explanatory variables is indicative of the highly variable flow regime observed during the summer sampling session. Summer flows result in significant water elevation changes, in which lower velocity areas near the edge of the channel become deeper, with faster velocities. During the fall, the river fluctuations are less, and result in more predictable responses, as observed by the higher proportion of variance typically explained by statistical models during fall periods.

In broader terms, modeling data suggests that the benthic community composition varied across the channel. Benthic biomass was likely greatest along the edge of the channel, and decreased slightly with depth and diminishing light intensity towards the thalweg. Benthic diversity was greatest along shallow channel areas closer to the wetted edge with lower velocities, adjacent to the interface between areas of laminar flow and small back eddies at the edge of the river. In the faster, deeper areas with increased cobble substrate, more sensitive taxa such as EPT were more prevalent. In contrast, in shallower, low velocity areas, or areas with finer substrates that were more depositional in nature, the benthic community consisted of more tolerant species such as Dipterans. Flows and shifts in flows were likely responsible for moving the wetted edge and shifting these communities, however flow was only a moderately reliable predictor of benthic communities in LCR.

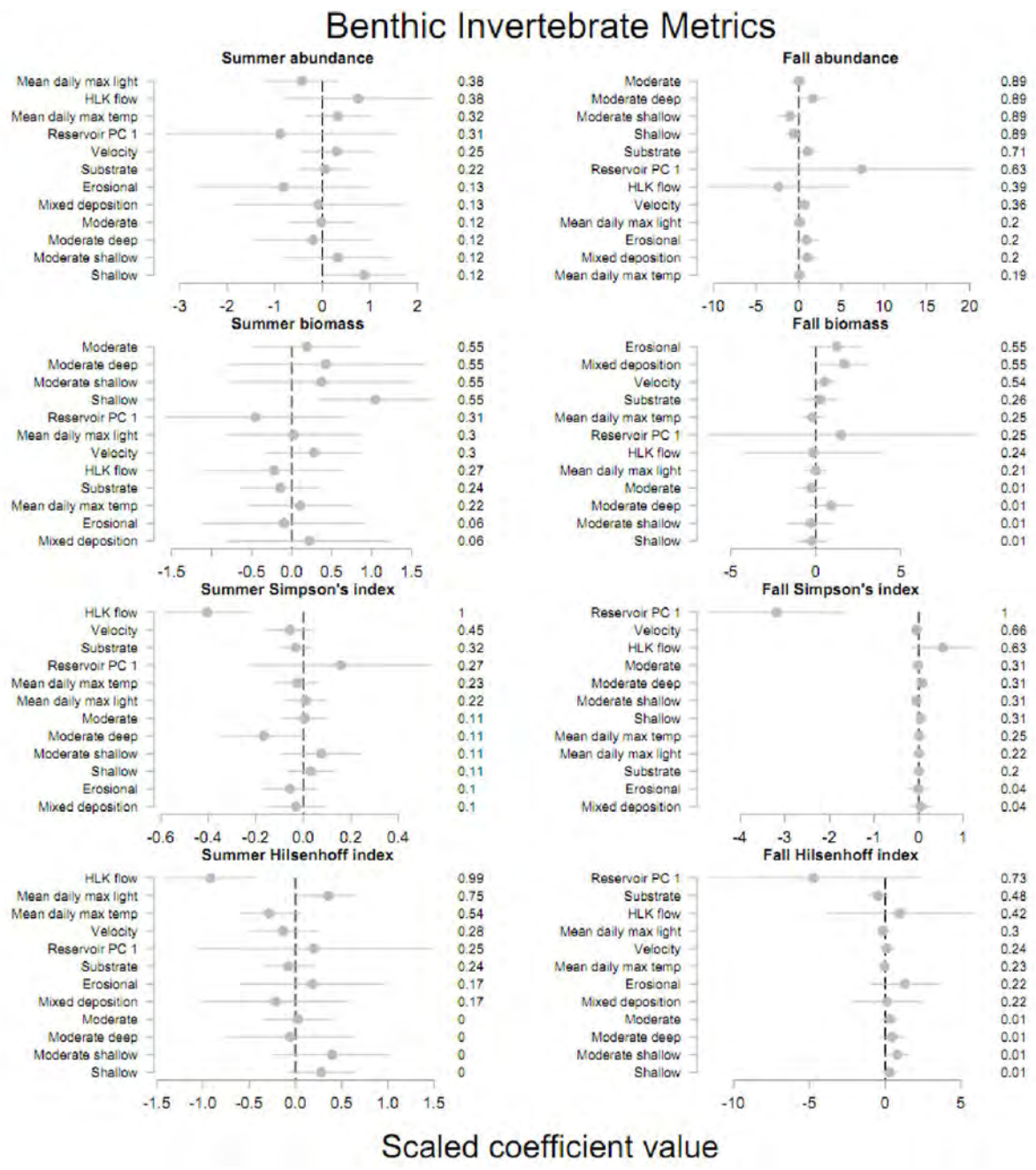

Figure 3-25: Scaled and Centered Parameter Estimates (*circles*) with 95% Unconditional Confidence Intervals (*lines*) from Averaged Predictive Linear Mixed-effects Models of Benthic Invertebrate Productivity and Diversity in the Lower Columbia River. Benthic responses included abundance, biomass, Simpson's index, and Hilsenhoff index. Explanatory variables included Mean Daily Maximum Water Temperature, Site Type (Erosional, Depositional, or Mixed), Substrate, Velocity, Mean Daily Maximum Light Intensity, HLK Flow, the first principal component of an analysis of reservoir conditions that represented reservoir temperature and elevation, and depth (transect). 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. Parameters (indicated on left) are ordered for each response variable by their relative importance (indicated on right) to the averaged model on a scale of 0 to 1.

3.5 Fish Diets

The benthic baskets have been successful at sampling fish food organisms and the data is considered 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 in LCR, was assessed using similar modeling techniques to periphyton and benthic invertebrates.

Fish food was modelled using a fish food index. Food for fish decreased with increasing HLK flows and increased with increasing light during the summer. In the fall period, the availability of fish food increased with velocity and decreased with increasing substrate size (Figure 3-26).

At this point, the fish food index is still in development and needs to be updated to reflect the diversity of other fish species occurring in LCR since only fish species / life stages and fish stomach contents data from the MCR has been used to assign fish food preference.

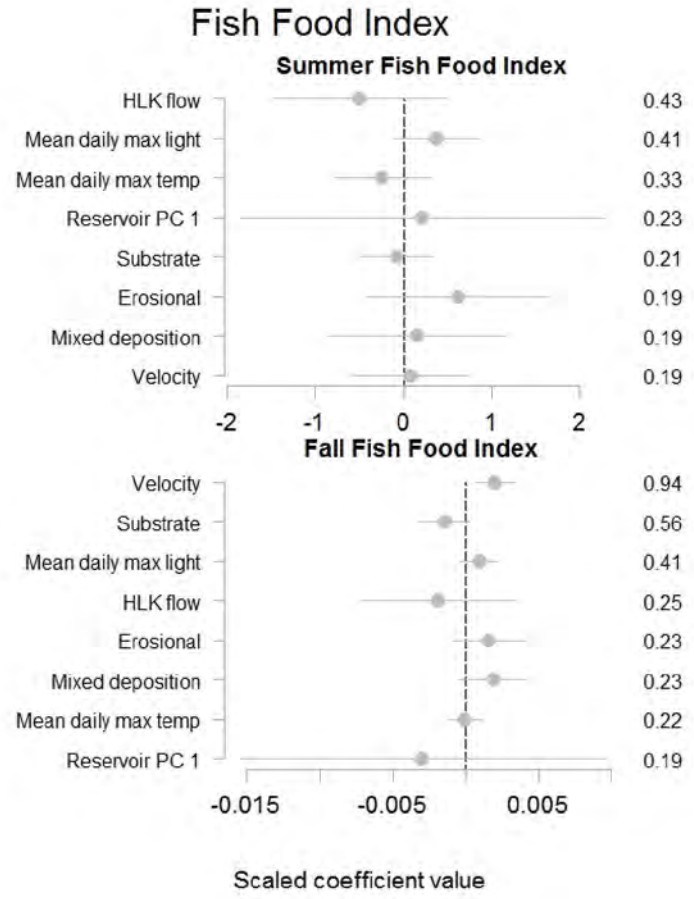

Figure 3-26: Scaled and Centered Parameter Estimates (circles) with 95% Unconditional Confidence Intervals (lines) from Averaged Predictive Linear Mixed-effects Models of a Fish Food Index in the Lower Columbia River. Explanatory variables included Mean Daily Maximum Water Temperature, Site Type (Erosional, Depositional, or Mixed), Substrate, Velocity, Mean Daily Maximum Light Intensity, HLK Flow, the first principle component of an analysis of reservoir conditions that represented reservoir temperature and elevation, and depth (transect). 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. Parameters (indicated on left) are ordered for each response variable by their relative importance (indicated on right) to the averaged model on a scale of 0 to 1.

4.0 DISCUSSION

4.1 Physical Habitat Monitoring

4.1.1 Water Temperature

Water temperatures at water quality index stations varied seasonally, ranging from approximately 4 to 18°C and this pattern was consistent among years. The seasonal patterns observed were similar across all index stations, although there may be a slight trend to later warming at stations downstream of the Kootenay River confluence.

The BC Ministry of Environment issues guidelines for water temperature in streams with known fish distributions. The guidelines state that water temperatures should be within ± 1°C of optimum temperature ranges for life history phases of the most sensitive fish species present (BCMOE 2012). The optimum temperature ranges of specific life history stages of species of interest are shown in Table 4-1. Data indicate some inconsistencies with recorded temperatures and the optimal temperature ranges of life history stages (Scofield et al. 2011).

Table 4-1: Optimum Temperature Ranges (°C) of Specific Life History Stages of Coldwater Species for Guideline Application (modified from BC MOE 2012)

Species	Incubation	Rearing	Migration	Spawning
Rainbow Trout	10.0 – 12.0	16.0 – 18.0	-	10.0 – 15.5
Mountain Whitefish	< 6.0	9.0 – 12.0	-	< 6.0

LCR water temperatures are 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 are variable and depend on season. Notably, flow does not appear to influence LCR temperature during the MWF flow period, but does seem to have a small effect during the RBT period when temperature increases with increasing flows. The statistical model describes a very high proportion of the variance, inferring that the identified factors are key parameters affecting river temperature. We therefore preliminarily accept hypothesis Ho_{1phy} that flow has the potential to affect river temperature, but the data suggests that other parameters such as air temperature, reservoir temperature, and to a lesser extent reservoir elevation must also be considered and that these factors are probably more important determinants of river temperature than flow or operating regimes during specific flow periods.

4.1.2 River Flows

The 2012 freshet peaked in the third week of July, with approximately 6,043.1 m^3/s recorded at the Birchbank Gauging Station. For comparison, the maximum mean daily river flows recorded in the first four years of this study (2008 – 2011) were 3,560.0, 2,730.0, 2,761.9 and 4,155.4 m^3/s, 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).

Since elevation data from before the implementation of either the MWF or RBT flow regimes are not available, modeling was used to predict river elevations for historic periods. This approach is appropriate because channel morphology has not changed significantly since 1984. Channel morphology can affect elevation at any given river cross section. For instance, 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 considered independently in modeling to ensure that site-specific effects and subsequent channel types would be apparent in the analysis.

The modeling data indicate that both of the post-implementation (1995 – 2007) and continued (2007 – 2012) MWF flow periods resulted in lesser changes in elevation between the spawning and incubation periods than pre-implementation of the flow regime(1984 – 1994). Further, the modelled elevations are very similar to those actually measured in the field corroborating our results. We therefore preliminary reject management sub-hypothesis HO_{2Aphy} and assume that the reduction in water level difference during the MWF flow period has likely been effective at reducing MWF egg dewatering during both post and continued implementation of the MWF flow period.

During the rainbow trout flow period, the modeling data indicate that both the post-implementation and the continued RBT flow regimes have resulted in 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 preliminary reject management sub-hypothesis HO_{2Aphy} and assume that the cumulative water declines during the RBT flow period have probably been effective at reducing RBT redd dewatering during both post and continued implementation of these flow periods.

4.1.3 Electrochemistry, Biologically Active Nutrients and Dissolved Oxygen Concentrations

The physical habitat monitoring management question #3 addresses the effects of the fish flow periods on electrochemistry and biologically active nutrients. Conductivity, TDS, alkalinity and pH were sampled to address electrochemistry, while the sampled biologically active nutrients included nitrate, ammonia, total phosphorus, and ortho-phosphate (SRP). Nutrients occurring as organic particulates require bacterial digestion before they are returned to a biologically active form, and of these, only total phosphorus was analyzed. The 2012 results were affected by the altered sampling frequency compared to the earlier years of this study.

The high 2011 freshet and the record high 2012 freshet flows lowered several electrochemistry parameters below the range reported from the 2008 – 2010 period with typical flows (Scofield et al. 2011). Specific conductance reacts to the concentration of dissolved solids in water, and therefore the record freshet likely diluted dissolved solids and resulted in lower conductivity readings. All electrochemistry parameters increased following freshet, when lower summer flows resulted in less dilution with a greater contribution from groundwater. As in many rivers, an inverse relationship was evident between flow and electrochemistry. Therefore, periods where flows are increased for fish should act to lower electrochemical parameters, but the influence of fish flows is very small compared to freshet. 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).

Unlike conductivity and hardness, pH was not measurably affected by the record 2011 and 2012 freshets. pH readings from 2012 were very similar to those reported during the first four years of this study (Scofield et al. 2011, Olson-Russello et al, 2012). Fish flows should therefore have a very minor influence on pH. All LCR pH values met the BC MoE Guideline and fell within LCR Objective range.

In any river system, there are numerous correlated influences on inorganic nutrient concentrations. The effects of the fish flow periods on biologically active nutrients were minor compared to the influences of the three major water sources. The nutrient status of the Lower Columbia River is heavily influenced by the limnology and nutrients status of the Arrow Lakes Reservoir (Hatfield, 2008) and of Kootenay Lake. Nutrients are added directly to LCR by numerous outfalls and by ALR fertilization. Finally, heavy freshets apparently contribute more inorganic nitrogen to the Columbia system than years with low runoff. For example, annual average inorganic nitrogen concentrations from 2011 and 2012 were approximately double the 2008 – 2010 values (Scofield et al. 2011). The importance of groundwater nutrient contributions are not known.

The forms of inorganic nitrogen include nitrate, ammonia and nitrite and these are key nutrients that are repeatedly consumed, transformed and released as water travels downstream. In LCR, inorganic nitrogen occurred as 77% nitrate and 23% ammonia during the 2011 sampling season and 100% nitrate in 2012. The higher 2012 flows may have maintained oxygen concentrations in the substrates and prevented the bacterial cycling of nitrate to the reduced forms while encouraging conversion to nitrate by nitrifying bacteria in the biofilm (periphyton) coating river substrates. Additionally increased flows excert more pressure on the substrates and limit the intrusion of groundwater which often has significant ammonia concentrations.

Unlike many rivers, nitrate concentrations remained elevated in the clear flow period, and may relate to the fertilization program on the Arrow Lakes Reservoir. Added nitrogen should arrive at LCR in July through October. During that period, inorganic nitrogen concentrations were highest at the sites closest to the dam, and diminished as the water flowed downstream, despite two municipal outfalls and Celgar adding inorganic and organic nutrients directly to LCR (Butcher, 1992). The Kootenay River displayed a similar inorganic nitrogen pattern to LCR during freshet, but nitrate concentrations dropped lower during clear flow period, further implying that the Arrow Lakes Reservoir maintains nitrate concentrations in LCR clear flows. All of LCR sites, the Kootenay and Norn's Creek were far below the BCMOE aquatic life nitrogen guidelines of 3 mg/L nitrate and 0.7 mg/L ammonia (BCMOE 2012).

Total phosphorus concentrations often trend with flow because more particulates are scoured into suspension. In LCR, total particulate phosphorus increased during the 2011 freshet, but remained below detection during the even larger 2012 freshet. The combination of dilution and reduced periphyton production may have produced the 2012 drop in T-P. If the observed response of LCR to freshets is broadly applicable, then moderately increased flows may improve the delivery of nutrients to periphyton while very high flows may prove deleterious to nutrient delivery. There is one key nutrient source that is independent of flows and that is from the lake fertilization programs. Phosphorus from

the Arrow Lakes Reservoir arrives in early fall as dissolved P and as algae cells exported to LCR.

The combination of these inorganic and organic forms of P did not induce excessive algae growth in LCR during the 2011 or 2012 growing season. However, strong periphyton growth did occur in the winter. LCR substrates may act as a temporary sink for dissolved inorganic phosphorus during the summer and fall, and release inorganic phosphorus during the winter as periphyton and aquatic macrophytes die back (Chaubey et al. 2007). Further, the Didymo mats sequester nutrients, making high concentrations available to the living layer of microflora on the top of the mat. Winter water quality sampling has commenced so this possibility can be assessed.

Ortho-phosphate never exceeded the detection limit of 0.01 mg/L in 2011 or 2012 LCR samples, with one exception – WQIS4, which is downstream of the Kootenay confluence and several outfalls. Kootenay T-P concentrations increased in 2011 and again in 2012, and these probably relate to the limnology and nutrient status of Kootenay Lake.

The 2012 LCR phosphorus results are all lower than the historic range recorded for Birchbank, and continue to follow a declining trend over the years (Holmes and Pommen 1999) as outfall water treatment improves. The recommended maximum soluble reactive phosphorus (SRP) component of total phosphorus to avoid excessive algae growth in rivers is 0.05 mg/L (Bowes et al. 2010) while the maximum recommended total phosphorus concentration is 0.03 mg/L (PWQO, 2005). Both ortho-phosphate and T-P concentrations were below these thresholds. Based on the 2011 and 2012 results, the Lower Columbia system is not in imminent danger of excessive algae growth, however there is ample phosphorus to maintain robust periphyton communities. The 2011 and 2012 total phosphorus concentrations would classify LCR as oligotrophic and the Kootenay as mesotrophic.

In light of the complex array of nutrient sources to LCR, it is not surprising that the influence of fish flows is not discernible. It is unlikely that fish flows will ever induce a change in biologically available nutrients that would result in a measurable change in periphyton growth. Conversely, the interplay between seasonal nutrient fluctuations from the ALR and from Kootenay Reservoir, deliberate nutrient introductions as fertilizer and the outfalls, the freshets, and possibly groundwater inputs will remain the dominant influences on the inorganic nutrients available to support periphyton growth in LCR. However, fish flows could affect the delivery of nutrients to periphyton during low flow periods particularly in the July to October window when ALR fertilized water is expected.

Nutrients influence the amount of dissolved oxygen produced by microflora. Dissolved oxygen levels directly affect MWF and RBT, and they always exceeded the 9 mg/L DO minimum guideline for the protection of aquatic life set by the BC Ministry of Environment (BCMOE 2012), and usually exceeded the 10 mg/L DO objective set for LCR (Butcher, 1992). The sites with the lowest DO saturation readings in 2012 were immediately downstream of HLK Dam (WQIS1 and WQIS2), and dissolved oxygen generally increased as the water travelled downstream. All collections were taken during daylight hours when photosynthesis generates DO, and not at night when respiration lowers DO (Doulos and Kindschi 1990). Like LCR, Kootenay River met the BCMOE DO guideline while a few measurements were lower than LCR objective set by the Province.

Given the data collected thus far, we have been unable to directly test whether MWF, RBT or FFF alter the availability of biological active nutrients and/or the electrochemistry of LCR. We expect the influence of fish flows on water quality to be subtle compared to the overwhelming effects of freshet, anthropogenic nutrient donation and photosynthesis within LCR. We anticipate that fish 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 LCR.

Future water quality sampling will be undertaken during the winter months to better address the MWF flow period hypothesis, and all water quality hypotheses will be addressed in future years when more data is available.

4.2 Periphyton Monitoring

The periphyton ecological monitoring management question HO_{2eco} addresses the effect of implementation of MWF, RBT and FF flows on total biomass accrual of periphyton in LCR.

To address this management question, periphyton monitoring completed in 2012/13 was concentrated in LCR Reach 2 at seven sites and in the most productive zones, ranging from the water's edge to 5 - 6m deep. Deeper thalweg areas can exceed 10 m depth and were not sampled. Based on limited research, it would appear that a correction factor of 1/2 is required to translate the artificial sampler results to the natural substrates of LCR. Further, there were no significant differences in abundance or biovolume between artificial substrates (closed cell Styrofoam and stone tile), suggesting that results from this project can be directly compared to the larger body of research utilizing stone tiles.

4.2.1 LCR Periphyton Community Structure

The LCR periphyton community is productive, diverse and variable. Most production metrics place LCR in the typical to productive categories (Table 4-2). Further, LCR periphyton community was more diverse and prolific than the MCR despite the greater annual discharge of LCR. Diatoms dominated the periphyton and there was no evidence of serious eutrophication of LCR. Upstream reservoirs donated diatoms that became incorporated into the LCR periphyton, accounting for as much as 10% of the total periphyton production. In other river systems, donated organisms can dominate the periphyton (Truelson and Warrington, 1994).

Table 4-2: Summary of typical LCR periphyton metrics from 2008 to 2013, 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 – 15.3
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	0.4 – 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	3.1
Organic matter (% of dry wt)		4 – 7%		1 – 10%	0.74%
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
Bacteria count water CFU/mL	0.1 – 10 x10^4	0.1 – 100 x10^5	2.4 x10^7	Not sampled	Not sampled
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	0.02 – 0.22

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.

The summer periods had the lowest overall productivity, while growth increased in the fall, but overall growth metrics were far higher in the winter. A flood year such as 2012 had the lowest overall productivity of the study years, particularly in the summer sampling period that was affected by RBT and freshet flows.

Key factors controlling LCR periphyton growth shifted seasonally from water temperature, substrate type, and the erosional or depositional character of a site during the high flow summer period, to substrate size, depth and light in the FFF period. Light was apparently a limiting factor governing periphyton growth in the fall and possibly in the winter but not in the summer. Winter water temperatures of 4 – 6^{o}C and short day length apparently exerted less influence than the benefits of stable flows, since the winter 2013 samplers produced the highest production yet recorded in LCR Reach 2. Cool winter water temperatures will restrict growth of most green algae and some cyanobacteria, but not diatoms or most flagellates (Wetzel 2001).

LCR periphyton community structure was always dominated by diatoms, as it is in most large rivers, however, large fluctuations occurred in the relative contributions of the other algae groups, notably the cyanobacteria and flagellates. Species richness was lowest in

the fall at the deep sites and highest in the summer at the shallow sites because higher flow conditions favored the proliferation of a small selection of tightly attached diatoms. Periphyton diversity was greatest in cobbled erosional areas at moderate depths during lower fall fluctuating flows.

Periphyton accrual was still increasing in the fall after 10 weeks, while accrual stabilized in just 4 – 6 weeks in the summer but at lower levels. The rapid development of a low peak biomass in the summer may relate to rapid cell loss through increased shear during freshet or from increased benthic invertebrate grazing.

Depositional areas are comparatively rare in LCR. They add a distinct range of periphyton and benthic invertebrate species not found at other sites. For example, decomposer organisms were more common in depositional sites, particularly in the summer when water temperatures of 10 -18^{o}C favored their growth (Wetzel, 2001). Periphyton diversity was greatest in depositional areas in summer during high flows but was low in the winter when sediment deposition hindered periphyton growth.

Flows and the related factors of velocity, shear, light penetration to the substrates and particulate suspension control periphyton growth. Extremely high flow events, such as those observed in summer 2012, reduced periphyton production in LCR. Reduced periphyton growth following high flow events is frequently observed in other river systems (Blinn et al 1995, Biggs 1996, Bunn and Arthington 2002).

4.2.2 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 influencing these production gradients are related to reservoir releases and the resultant flows. Like any river, the position of the peak velocity zones shift with flows. For example, the position of the interface between the main cool river flow and the shallow, warmer back-eddy zones is directly related to flow and it appears that LCR periphyton community is associated with this interface in the river. This suggests that flows have a direct influence on periphyton production and community in LCR. However, inferences on the effects of operating flow regimes during the MWF and RBT periods are difficult to make with confidence due to high variability in the statistical models.

Other influences on periphyton development may be important. For example, periphyton grazing by benthic invertebrates was evident on many samplers. Periphyton standing crop losses to grazing may be significant, particularly when water temperatures exceed 10^{o}C in the summer.

4.2.2.1 Spring/Summer RBT flows Apr 1 – Jun 30

In all the data collected thus far, summer high flow periods had the lowest periphyton production. High freshet flows and consequent higher velocities apparently increased shear to the point that the summer standing crop of periphyton was reduced. The results from summer 2012 with record flows support this theory. Its periphyton production was lower than production from the typical flow years of 2008 – 2010. Biggs (1996) found that differing disturbance frequencies was the dominant influence controlling average periphyton biomass. In his work and in LCR, high flows affected filamentous greens and Didymo more than the diatoms, causing shifts in community structure. This is because the

loss of filamentous periphyton is not a simple linear function of increased shear stress with the onset of increased velocities. Relatively large amounts of filaments can be dislodged from the stream bed with small increases in velocity, while the loss rate of tightly attached diatoms tends to increase linearly with increased shear stress (Biggs, 1996). Increased freshet flows during the RBT flow period (early May in 2013) were sufficient to shear off the Didymo filaments that had developed during stable winter flows. Periphyton production at depositional sites included a distinct species assemblage and was less affected by high spring/summer flows than the periphyton at erosional sites.

Since freshet flows overshadow the RBT flows, we tentatively accept the null hypothesis HO_{2Beco} that RBT flows do not increase total biomass accrual of periphyton.

4.2.2.2 Fall fluctuating flows Sept 1 – Oct 31

Across all years, periphyton productivity increased during the fall at most sampled depths, with the exception of several shallow sites. Periodic dewatering of substrates at the water's edge likely reduced production at the shallowest substrates during the fall. Dewatering losses along peripheral substrates is progressive. The longer substrates remain dewatered, the greater the periphyton losses become (Schleppe *et al*. 2013). 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, and this banding pattern was similar to banding patterns observed in the MCR. Fall periphyton production also declined at deep locations as a function of growth restriction by light penetration and increasing velocity.

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 (Schindler et al. 2009). The arrival of this surge in drift algae occurs at a time when flows are moderate, which allows more of the lake forms to join the periphyton (Bonnett et al., 2009). Lake forms accounted for up to 10% of the observed periphyton in the fall, and increased overall periphyton diversity.

Overall, the mean daily flows during the FFF period were generally quite stable. These moderate flows allowed more periphyton growth compared to the summer. We preliminarily accept the null hypothesis HO_{2Ceco} that fall fluctuating flows do not increase total biomass accrual of periphyton, since the fall biomass data was typical of large rivers.

4.2.2.3 Winter MWF flows Jan 1 – Mar 31

Winter periphyton growth was far higher than the preceding seasons and it included proportionately more low light tolerant cyanobacteria. The erosional sites benefitted the most from the low, stable winter flows, while settling sediments limited production at the depositional sites (e.g., Site 6).

Stable LCR winter flows permitted the thickest growth of the invasive Didymo seen in the annual cycle. The alteration of 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 helps explain the high predominance of Didymo observed during the winter sampling session.

In light of these findings, we preliminarily reject hypothesis HO_{2Aeco} that MWF flows do not increase total accrual of periphyton biomass. Weekly measurements of chl-a accrual are planned for the MWF flow period in 2014 to provide additional information that will help address this hypothesis.

4.2.2.4 Summary of Flow Related Effects on Periphyton Community and Production

Flow is an important determinant of the periphyton community. Flow affects most physical habitat conditions, such as light, velocity and shear, which are shown to be important factors affecting periphyton production and community. Thus, we can confidently conclude that flows in LCR affect periphyton community structure and productivity. However, the specific effects of the MWF and RBT managed flow regimes are difficult to determine in light of larger-scale flow effects, such as freshet. Future iterations of our statistical models will help elucidate the specific effects of these flow regimes on periphyton production and community structure.

4.2.3 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. Additionally, drifting algae from reservoir releases also provides food. Drift forage quality was lower than the periphyton quality, but was still important to filter feeders. The amount of chlorophyll-a observed in the drift increased as water travelled downstream through LCR, increasing the food supply for filter feeders.

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 such as *Ulothrix*, and *Cladophora* may not be directly edible, but they create microhabitats that can harbor 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 extensive proliferation of Didymo. The muco-polysaccharide Didymo filaments can be problematic due to their resistance to grazing by 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 winter stable low flows, then the stable winter low flows may be deleterious to benthic invertebrate development in the mid-depth cobble substrates that become coated with the Didymo filament mats.

4.3 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 four years of benthic invertebrate data have been collected between 2008 and 2013. 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.

4.3.1 Benthic Invertebrate Community Structure

Table 4-3 provides a comparison of benthic invertebrates in different river systems. The benthic community in LCR is remarkably more stable, diverse and productive compared to MCR below the Revelstoke Dam. This is apparent when comparing the mean number of invertebrates per sample. The dearth of individuals found in MCR is thought to be due to daily fluctuations in flow that result in regular exposure of the river channel (Schleppe *et al.* 2013). Despite the higher average annual discharge of LCR, its more consistent 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 favors the proliferation of orthoclad chironomids (Munn and Brusven 1991). In LCR, chironomids were top contributors to relative abundance in 2008 – 2011 (Scofield et al. 2011), and continued to do so in 2012 and 2013. An increased predominance of filter feeding benthic invertebrates has also been documented in regulated river systems (2009); and LCR has high relative abundances of web spinning caddisflies of the family Hydropsychidae. Thus, in some aspects, 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 contaminants (Celgar), 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 absolute certainty.

Table 4-3: 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 (Spences 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. The 2008 – 2012 modeling data suggests that densities were greatest during periods of stable flow in moderate depth areas. High peak flows as observed in 2012 appeared to reduce benthic diversity, and to a lesser extent abundance and biomass. Although winter data was not included in the model, we suspect that the stable winter flows were critical to the high abundance and biomass documented for periphyton and invertebrate communities, as well as for the overriding success of Didymo.

The more sensitive taxa, such as EPT, were more abundant on 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.

Although it is difficult to attribute causal effects to specific flow regulation regimes, we speculate that implementation of the MWF and RBT flow regimes does affect the benthic

invertebrate communities in LCR, rejecting the null hypotheses. For example, during the MWF period, flows were generally low and stable, which may have contributed to the proliferation of Didymo and effectively altered benthic periphyton and invertebrate production on cobble substrates.

4.3.2 Winter MWF Flows

Sampling in 2012/13 included the first event of winter sampling and also coincided with a year of unprecedented freshet flows. The winter abundance and biomass data appeared similar to summer and fall data from earlier sampling years. The percent EPT was the only diversity measure that was consistently lower in winter compared to other sampling seasons (14 compared to 59 and 34 % in summer and fall, respectively). Despite the lower percentage of EPT, EPT richness was comparable to previous years suggesting that there was still a diverse assemblage of more pollution sensitive species present, but at lower abundances during the winter. In contrast, percent Chironomidae tended to be higher during winter (48% compared to 29 and 45% in the summer and fall). 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). The decline in EPT and enhanced Chironomidae during the winter may coincide with the elevated presence of Didymo. Shelby (2006) reported a reduction in the number of different invertebrate taxa when Didymo was present. We also documented this trend as the maximum species richness during winter was 29, compared to an average maximum species richness of 44.5 in the summer and 36 in the fall. Further, Didymo was most extensive at moderate depths, while the invertebrate maximum species richness was lowest at the sites of moderate depth.

Because there has only been one winter sampling session, it is difficult to extrapolate the outlined 2013 trends to future winters, particularly given the high flow events documented in 2012. 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 single year of data, this did not appear to be true. However, the winter samplers were left in the river for 12 weeks compared to between 6 to 11 weeks for previous 2008 – 2012 summer and fall sessions. The longer deployment 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.

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 preliminarily reject the null hypothesis that the continued implementation of MWF flows does not affect the biomass, abundance and composition of benthic invertebrates in LCR.

4.3.3 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. Our modeling data during the summer sampling period suggests that freshet is a more predominant feature, but does provide indication that flows, particularly high flows, are also likely affecting the benthic invertebrate community. Despite this, we still hypothesize that the reduction in substrate dewatering during the RBT flow period has acted to stabilize flows and the invertebrate 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.

4.3.4 Fall Fluctuating Flows

During the FFF period, flows are actually quite stable. Mean daily variation in flows was less than that observed during the summer period, noting that hourly flow variation (i.e., water elevation within a given day) was not investigated. 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.

At this time, we preliminarily reject all four null hypotheses because at minimum, flow management has resulted in changes to the LCR benthic invertebrate community. In future years, we will attempt to elucidate the specific effects of the MWF, RBT, and FFF periods on the benthic community as more data is acquired.

4.4 Food for Fish

The availability of food for fish was affected by season, similar to the patterns identified for all benthic invertebrate abundance and diversity in LCR. Modeling data suggest that high peak flows during summer periods, particularly in 2012, had an overall negative effect on food for fish. However, during more stable flow periods, food availability for fish was positively associated with velocity. This suggests that during the RBT flow period, high peak freshet flows will reduce food availability for fish, making detection of specific effects associated with the RBT flow regime on fish food availability more difficult.

During the FFF period, the availability of food for fish was greatest in areas of higher velocity. Areas of higher velocity were more typical of erosional, cobble banks, which tended to have greater predominance of the more sensitive EPT taxa. Although not

specifically modelled, we hypothesize that fish food availability will decrease in the winter due to the establishment of Didymo and less favourable habitat conditions. The winter data suggests that Chironomid taxa were more prevalent in shallow locations where Didymo establishment was greater and EPT taxa were more prevalent in deeper, mid channel areas where higher flows reduced Didymo densities.

5.0 RECOMMENDATIONS

1. Given our observation that Didymo mats adversely affect periphyton and benthic invertebrate communities, especially during the winter sampling period, it would be prudent to establish a Didymo monitoring protocol that could effectively classify the extent of Didymo during each productivity sampling period. Didymo data should be collected in such a way that it can be incorporated into future modeling. If in the future it can be confirmed that Didymo is in fact playing a role in altered community structure, then intentional flushing of the system with higher flows during the MWF flow period, may be beneficial to control Didymo mat development.

2. There has been variation in the duration of sampler deployments in the summer, fall and winter sampling sessions. Future sampling should attempt a consistent sampling duration of 10 weeks across all seasons.

3. Total phosphorus has been regularly sampled over the study years but total N sampling has not. Adding total N to the analysis would provide additional information on nutrient cycling, particularly the contribution of nitrogen in organic forms, at a small cost.

4. Benthic invertebrate time series sampling should occur to document accrual of benthic invertebrates. Understanding benthic accrual will be necessary to understand differences between deployment times during the summer, fall, and winter sampling periods.

5. A study should be designed to understand the specific effects of spring freshet on benthic communities. Samplers could be deployed prior to freshet and allowed to achieve a stable community (approximately 5 to 6 weeks). Sampling could occur before and after freshet and the two periods compared as paired samples to better understand the effects of freshet and how benthic communities are affected by RBT flows.

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Appendix A1. Water Quality Data Collected in LCR during 2012.

Sample Location	Date	Hardness	Alkalinity	TDS	TSS	Turbidity	Ammonia as N	Nitrate+Nitrite as N	Nitrate as N	Nitrite as N	TDN	T-Phosphorous	Ortho as P	T-Recoverable Mg	T-Recoverable P
		mg/L	mg/L	mg/L	mg/L	NTU	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
RDL		5	1	5	1	0.1	0.02	0.02	0.01	0.01	0.01	0.002	0.01	2	0.1
WQIS1	02-Jun-12	63.6		41	<1	0.5	<0.020	0.086	0.086	<0.010	0.086	<0.002	<0.01	18	4.5
WQIS2	2-Jun-12	64.1		50	<1	0.5	<0.020	0.091	0.091	<0.010	0.091	<0.002	<0.01	18	4.5
WQIS3	2-Jun-12	56.4		43	<1	0.5	<0.020	0.072	0.072	<0.010	0.072	<0.002	<0.01	16	4
WQIS4	2-Jun-12	66.2		67	<1	0.8	<0.020	0.095	0.095	<0.010	0.095	<0.002	<0.01	19	4.5
WQIS5	2-Jun-12	67.5		66	<1	0.9	<0.020	0.096	0.096	<0.010	0.096	<0.002	<0.01	19	4.7
Kootenay R.	2-Jun-12	68.7		53	<1	0.8	<0.020	0.095	0.095	<0.010	0.095	<0.002	<0.01	< 2	0.4
Norns Ck	2-Jun-12	<5.0		<5	15	4.1	<0.020	<0.020	<0.010	<0.010	<0.010	0.02	<0.01	20	4.8
WQIS1	14-Aµg-12		45	65	1	0.6	<0.020	0.11	0.110	<0.010	0.110	0.010	<0.01		
WQIS2	14-Aµg-12			46	<1	0.5	<0.020	0.114	0.114	<0.010	0.114	0.010	<0.01		
WQIS3	14-Aµg-12			58	1	0.4	<0.020	0.093	0.093	<0.010	0.093	0.010	<0.01		
WQIS4	14-Aµg-12			68	2	0.5	<0.020	0.088	0.088	<0.010	0.088	0.020	0.02		
WQIS5	14-Aµg-12		51	70	2	0.5	<0.020	0.098	0.098	<0.010	0.098	0.010	<0.01		
Kootenay R.	14-Aµg-12		61	87	2	0.7	<0.020	0.074	0.074	<0.010	0.074	0.020	<0.01		
Norns Ck	14-Aµg-12			10	<1	0.3	<0.020	<0.020	<0.010	<0.010	<0.010	0.020	<0.01		
WQIS1	25-Oct-12	49		52	<1	0.3	<0.020	0.102	0.102	<0.010	0.102	0.005	<0.01		
WQIS2	25-Oct-12			70	<1	0.3	<0.020	0.095	0.095	<0.010	0.095	0.009	<0.01		
WQIS3	25-Oct-12			56	<1	0.3	<0.020	0.096	0.096	<0.010	0.096	0.006	<0.01		
WQIS4	25-Oct-12			58	<1	0.4	<0.020	0.085	0.085	<0.010	0.085	0.004	<0.01		
WQIS5	25-Oct-12	52		56	<1	0.4	<0.020	0.082	0.082	<0.010	0.082	0.006	<0.01		
Kootenay R.	25-Oct-12	64		70	<1	0.6	<0.020	0.038	0.038	<0.010	0.038	0.011	<0.01		
Norns Ck	25-Oct-12			26	<1	0.2	<0.020	<0.020	0.018	<0.010	0.018	0.005	<0.01		

Appendix A2. Water Quality Data Collected in LCR using a Hand-held Metre during 2012.

Reach	Station	Waterbody	Date	°C	pH	DO %	DO ppm	Cond µS/cm	TDS ppm	Salinity
R1	WQIS1	Columbia	01-Jun-12	10.9	7.87	112.6	11.49	58.72	33.33	0.07
R1	WQIS2	Columbia	01-Jun-12	10.4	7.79	113.8	11.75	60.00	34.30	0.07
R2	WQIS3	Columbia	01-Jun-12	9.39	7.53	115.5	12.23	49.79	28.50	0.06
R3	WQIS4	Columbia	01-Jun-12	9.61	7.79	119.6	12.61	64.26	36.71	0.07
R3	WQIS5	Columbia	01-Jun-12	9.63	7.66	118.1	12.46	64.68	36.71	0.07
R2	WQ C2	Kootenay	01-Jun-12	9.41	7.56	124.5	13.19	67.23	38.16	0.08
R2	WQ C1	Norns	01-Jun-12	4.93	7.77	111.4	13.15	8.94	4.83	0.01
R1	WQIS1	Columbia	14-Aµg-12	15.3	7.82	105.7	9.82	100.85	57.00	0.11
R1	WQIS2	Columbia	14-Aµg-12	15.7	7.81	108.5	10	105.53	59.90	0.12
R2	WQIS3	Columbia	14-Aµg-12	15.6	7.83	110.6	10.22	111.06	62.80	0.12
R3	WQIS4	Columbia	14-Aµg-12	16.4	7.89	110.2	10.04	113.19	64.25	0.13
R3	WQIS5	Columbia	14-Aµg-12	16.5	8.04	110.3	10.07	116.17	65.70	0.13
R2	WQ C2	Kootenay	14-Aµg-12	18.1	8.15	109.1	9.57	125.53	71.50	0.14
R2	WQ C1	Norns	14-Aµg-12	13.5	8.6	93.1	9.04	77.02	43.96	0.09
R1	WQIS1	Columbia	25-Oct-12	11.5	7.63	95.6	9.75	105.53	59.90	0.12
R1	WQIS2	Columbia	25-Oct-12	11.4	7.63	95.9	9.79	105.11	59.90	0.12
R2	WQIS3	Columbia	25-Oct-12	11.3	7.63	94.9	9.7	106.38	60.39	0.12
R3	WQIS4	Columbia	25-Oct-12	11.5	7.61	95	9.68	108.94	61.84	0.12
R3	WQIS5	Columbia	25-Oct-12	11.5	7.93	95.1	9.68	106.38	60.39	0.12
R2	WQ C2	Kootenay	25-Oct-12	11.9	7.62	95	9.59	121.28	68.60	0.14
R2	WQ C1	Norns	25-Oct-12	5.36	8.33	97.7	11.57	96.17	54.59	0.11

Appendix A3. Periphyton Summary Tables (2008 to 2013).

Table A3-1: Relative abundance and relative bio volume of periphyton taxa in the fall of each year from 2008 to 2012.

Taxa	2008 Relative Abundance (%)	2008 Relative Bio Volume (%)	Taxa
Micro/Picoflagellates	21.11	22.81	Cymbellaceae Didymosphenia
Achnanthaceae Achnanthidium	5.23	7.06	Fragilariaceae Fragilaria
Fragilariaceae Fragilaria	4.28	5.76	Ulotrichaceae Ulothrix
Fragilariaceae Staurosira	3.27	3.76	Fragilariaceae Synedra
Synechococcacea Synechococcus sp.	2.18	3.7	Fragilariaceae Tabellaria
Chroococcaceae Anacystis	1.87	1.41	Achnanthaceae Cocconeis
Fragilariaceae Synedra	1.26	1.26	Cymbellaceae Cymbella
Cymbellaceae Didymosphenia	1.23	1.1	Rhopalodiaceae Epithemia
Oscillatoriaceae Planktolyngbya	1.15	0.99	Fragilariaceae Diatoma
Gomponemaceae Gomphonema	1.01	0.71	Gomponemaceae Gomphonema

Taxa	2009 Relative Abundance (%)	2009 Relative Bio Volume (%)	Taxa
Achnanthaceae Achnanthidium	6.81	15.59	Cymbellaceae Didymosphenia
Fragilariaceae Fragilaria	4.31	6.18	Ulotrichaceae Ulothrix
Unidentified coccoid green	2.68	5.25	Fragilariaceae Fragilaria
Fragilariaceae Staurosira	2.19	4.72	Fragilariaceae Tabellaria
Fragilariaceae Tabellaria	1.9	4.69	Fragilariaceae Synedra
Gomponemaceae Gomphonema	1.76	1.5	Gomponemaceae Gomphonema
Cymbellaceae Cymbella	1.52	1.08	Achnanthaceae Eucocconeis
Fragilariaceae Synedra	1.31	1.07	Cymbellaceae Cymbella
Fragilariaceae Staurosirella	1.13	0.99	Fragilariaceae Diatoma
Cymbellaceae Didymosphenia	0.94	0.95	Zygnemataceae Moμgeotia

Taxa	2010 Relative Abundance (%)	2010 Relative Bio Volume (%)	Taxa
Fragilariaceae Fragilaria	12.84	18.18	Fragilariaceae Fragilaria
Achnanthaceae Achnanthidium	7.49	3.9	Ulotrichaceae Ulothrix
Fragilariaceae Staurosira	1.85	3.1	Fragilariaceae Synedra
Coscinodiscaceae Cyclotella	1.74	2.34	Fragilariaceae Diatoma
Fragilariaceae Diatoma	1.38	1.78	Fragilariaceae Tabellaria
Unidentified coccoid green	0.85	1.49	Coscinodiscaceae Cyclotella
Gomponemaceae Gomphonema	0.67	1.17	Cymbellaceae Cymbella
Fragilariaceae Fragilariforma	0.62	1.11	Gomponemaceae Gomphonema
Fragilariaceae Staurosirella	0.58	0.79	Achnanthaceae Achnanthidium
Fragilariaceae Tabellaria	0.57	0.66	Desmidiaceae Cosmarium

Taxa	2012 Relative Abundance (%)	2012 Relative Bio Volume (%)	Taxa
Merismopediaceae Synechocystis	7.27	6.64	Fragilariaceae Tabellaria
Achnanthaceae Achnanthidium	7.14	6.07	Cymbellaceae Didymosphenia
Oscillatoriaceae Planktolyngbya	6.61	5.53	Fragilariaceae Synedra
Cyanobacteriaceae Aphanothece	6.22	3.13	Fragilariaceae Fragilariforma
Chromulinaceae Ochromonas	2.74	2.66	Achnanthaceae Eucocconeis
Unidentified flagellates	2.46	2.31	Chromulinaceae Ochromonas
Cyanobacteriaceae Aphanocapsa	2.22	2.03	Naviculaceae Frustulia
Fragilariaceae Fragilariforma	1.9	1.69	Achnanthaceae Cocconeis
Fragilariaceae Staurosira	1.67	1.41	Oedogoniaceae Bulbochaete
Fragilariaceae Synedra	1.67	1.29	Phacaceae Phacus

Table A3-2: Relative abundance and relative bio volume of periphyton taxa in the summer of each year from 2008 to 2012.

Taxa	2008 Relative Abundance (%)	2008 Relative Bio Volume (%)	Taxa
Micro/Picoflagellates	46.54	5.9	Fragilariaceae Tabellaria
Synechococcacea Synechococcus sp.	6.42	4.9	Ulotrichaceae Ulothrix
Achnanthaceae Achnanthidium	3.26	4.76	Fragilariaceae Fragilaria
Merismopediaceae Synechocystis	2.86	4.73	Cymbellaceae Didymosphenia
Fragilariaceae Staurosira	1.78	4.17	Achnanthaceae Cocconeis
Oscillatoriaceae Planktolyngbya	1.31	3.2	Desmidiaceae Cosmarium
Fragilariaceae Fragilaria	1.23	2.33	Fragilariaceae Synedra
Chroococcaceae Anacystis	0.96	2.24	Scenedesmaceae Coelastrum
Achnanthaceae Cocconeis	0.95	1.59	Gomponemaceae Gomphonema
Merismopediaceae Merismopedia	0.63	1.56	Rhopalodiaceae Epithemia

Taxa	2009 Relative Abundance (%)	2009 Relative Bio Volume (%)	Taxa
Achnanthaceae Achnanthidium	9.4	7.31	Fragilariaceae Tabellaria
Fragilariaceae Staurosira	3.4	4.86	Cymbellaceae Didymosphenia
Fragilariaceae Fragilaria	2.83	4.5	Fragilariaceae Fragilaria
Unidentified coccoid green	1.69	3.65	Ulotrichaceae Ulothrix
Fragilariaceae Tabellaria	1.18	3.6	Fragilariaceae Synedra
Fragilariaceae Staurosirella	1.15	1.61	Achnanthaceae Eucocconeis
Coscinodiscaceae Cyclotella	1.05	1.43	Amphipleuraceae Frustulia
Amphipleuraceae Frustulia	0.97	1.24	Coscinodiscaceae Cyclotella
Cymbellaceae Cymbella	0.85	1.22	Achnanthaceae Cocconeis
Achnanthaceae Cocconeis	0.75	1.16	Cymbellaceae Cymbella

Taxa	2010 Relative Abundance (%)	2010 Relative Bio Volume (%)	Taxa
Achnanthaceae Achnanthidium	10.03	10.86	Fragilariaceae Fragilaria
Fragilariaceae Fragilaria	5.44	6.59	Ulotrichaceae Ulothrix
Unidentified coccoid green	3.3	5.59	Eremosphaeraceae Eremosphaera
Fragilariaceae Staurosira	2.59	2.71	Desmidiaceae Cosmarium
Prasiolaceae Stichococcus	1.92	2.2	Fragilariaceae Tabellaria
Coscinodiscaceae Cyclotella	0.99	2.04	Fragilariaceae Synedra
Chroococaceae Chroococcus	0.95	1.65	Coscinodiscaceae Cyclotella
Fragilariaceae Staurosirella	0.81	1.64	Achnanthaceae Cocconeis
Pseudanabaenaceae Pseudanabaena	0.69	1.48	Achnanthaceae Achnanthidium
Dinobryaceae Chrysococcus	0.66	1.18	Gomponemaceae Gomphonema

Taxa	2012 Relative Abundance (%)	2012 Relative Bio Volume (%)	Taxa
Oscillatoriaceae Planktolyngbya	20.75	4.67	Fragilariaceae Synedra
Chroococcaceae Anacystis	9.52	4.38	Fragilariaceae Tabellaria
Cyanobacteriaceae Aphanothece	8.5	4.2	Naviculaceae Frustulia
Merismopediaceae Synechocystis	5.34	3.74	Achnanthaceae Eucocconeis
Achnanthaceae Achnanthidium	5.26	3.57	Fragilariaceae Fragilariforma
Unidentified flagellates	1.63	3.37	Cymbellaceae Didymosphenia
Fragilariaceae Fragilariforma	1.58	2.29	Ulotrichaceae Ulothrix
Fragilariaceae Staurosira	1.07	2.25	Cyanobacteriaceae Aphanothece
Fragilariaceae Synedra	0.94	1.9	Cymbellaceae Cymbella
Oscillatoriaceae Oscillatoria	0.91	1.47	Achnanthaceae Achnanthidium

Table A3-3: Relative abundance and relative bio volume of periphyton taxa in the winter of 2013.

Taxa	2013		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Oscillatoriaceae Oscillatoria	14.67	28.35	Cymbellaceae Didymosphenia
Fragilariaceae Fragilariforma	7.35	4.98	Fragilariaceae Fragilariforma
Merismopediaceae Synechocystis	6.93	4.65	Fragilariaceae Synedra
Oscillatoriaceae Planktolyngbya	6.74	4.55	Fragilariaceae Diatoma
Fragilariaceae Diatoma	4	3.89	Ulotrichaceae Ulothrix
Gomphonemataceae Gomphoneis	3.55	2.61	Gomphonemataceae Gomphoneis
Achnanthaceae Achnanthidium	2.73	1.41	Cymbellaceae Cymbella
Unidentified flagellates	1.87	1.04	Cymbellaceae Cymbopleura
Chromulinaceae Ochromonas	1.43	0.96	Cladophoraceae Cladophora
Cymbellaceae Cymbella	1.27	0.83	Fragilariaceae Tabellaria

Table A3-4: Relative abundance and relative bio volume of periphyton broad taxanomic groups in the fall of 2012.

Taxa	2008		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	70.71	92.33	Diatoms
Flagellates	21.98	6.82	Green Algae
Blue-Green Algae	6.67	0.70	Flagellates
Green Algae	0.64	0.15	Blue-Green Algae
Dinoflagellates	0.00	0.00	Dinoflagellates
Taxa	2009		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	93.70	91.03	Diatoms
Green Algae	4.16	8.68	Green Algae
Blue-Green Algae	1.73	0.22	Dinoflagellates
Flagellates	0.27	0.06	Flagellates
Dinoflagellates	0.15	0.01	Blue-Green Algae
Taxa	2010		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	96.60	93.66	Diatoms
Green Algae	1.71	6.03	Green Algae
Blue-Green Algae	1.27	0.15	Flagellates
Flagellates	0.37	0.14	Dinoflagellates
Dinoflagellates	0.05	0.02	Blue-Green Algae
Taxa	2012		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	65.54	90.83	Diatoms
Blue-Green Algae	18.99	4.47	Flagellates
Flagellates	8.61	4.12	Green Algae
Green Algae	6.78	0.37	Blue-Green Algae
Dinoflagellates	0.08	0.21	Dinoflagellates

Table A3-5: Relative abundance and relative bio volume of periphyton broad taxanomic groups in the summer of 2012.

Taxa	2008		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Flagellates	47.59	84.32	Diatoms
Diatoms	37.84	11.34	Green Algae
Blue-Green Algae	12.83	3.25	Flagellates
Green Algae	1.69	0.64	Blue-Green Algae
Dinoflagellates	0.05	0.45	Dinoflagellates
Taxa	2009		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	96.99	95.41	Diatoms
Green Algae	2.16	4.37	Green Algae
Blue-Green Algae	0.73	0.11	Flagellates
Flagellates	0.08	0.10	Dinoflagellates
Dinoflagellates	0.04	0.01	Blue-Green Algae
Taxa	2010		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	88.98	81.06	Diatoms
Green Algae	6.65	17.81	Green Algae
Blue-Green Algae	3.16	0.67	Flagellates
Flagellates	1.09	0.41	Dinoflagellates
Dinoflagellates	0.11	0.06	Blue-Green Algae
Taxa	2012		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	46.57	90.61	Diatoms
Blue-Green Algae	39.84	5.07	Green Algae
Green Algae	8.95	2.23	Flagellates
Flagellates	4.37	1.28	Dinoflagellates
Dinoflagellates	0.27	0.81	Blue-Green Algae

Table A3-6: Relative abundance and relative bio volume of periphyton broad taxanomic groups in the winter of 2013.

Taxa	2013		Taxa
	Relative Abundance (%)	Relative Bio Volume (%)	
Diatoms	64.38	92.76	Diatoms
Blue-Green Algae	29.80	6.10	Green Algae
Flagellates	4.29	0.59	Flagellates
Green Algae	1.54	0.55	Blue-Green Algae
Dinoflagellates	0.00	0.00	Dinoflagellates

Table A3-7: Summary statistics for periphyton in the Lower Columbia River in the fall 2012. Samplers occurred at six locations of increasing relative depth (Shallow to Deep).

Diversity Measures	Statistics	Shallow	Moderate Shallow	Moderate	Moderate Deep	Deep
Abundance (# /$cm^3 \times 10^5$)	Mean (±SD)	3.5 ± 1.72	8.69 ± 3.11	7.34 ± 2.84	5.66 ± 3.88	4.23 ± 2.04
	Median	2.8	8.25	6.72	6.3	3.66
	Minimum	1.89	5.47	4.52	0.6	2.3
	Maximum	6.21	13.81	11.65	10.95	6.62
Species Richness (#)	Mean (±SD)	26.6 ± 3.29	30 ± 2.53	31.17 ± 2.79	26.6 ± 9.66	29.8 ± 4.02
	Median	26	30.5	31	27	30
	Minimum	24	27	28	12	25
	Maximum	32	33	36	39	36
Bio Volume (cm^3/m^2)	Mean (±SD)	0.73 ± 0.29	3.2 ± 1.47	2.55 ± 2.43	1.49 ± 1.36	1.62 ± 1.15
	Median	0.56	2.94	1.97	1.22	1.2
	Minimum	0.48	1.54	0.71	0.1	0.64
	Maximum	1.11	5.54	7.34	3.64	3.61
Simpson's Index	Mean (±SD)	0.84 ± 0.07	0.82 ± 0.04	0.86 ± 0.04	0.84 ± 0.02	0.82 ± 0.06
	Median	0.87	0.82	0.86	0.84	0.84
	Minimum	0.74	0.75	0.8	0.81	0.75
	Maximum	0.91	0.86	0.9	0.86	0.88
Chlorophyll-a ($\mu g/m^2$)	Mean (±SD)	1.38 ± 0.42	2.11 ± 0.74	2.16 ± 0.85	2.42 ± 1.33	2.29 ± 1.04
	Median	1.5	1.98	2.36	2.5	1.98
	Minimum	0.74	1.24	1.09	0.73	1.21
	Maximum	1.89	3.36	3.29	3.74	3.65
Autotrophic Index	Mean (±SD)	6106.92 ± 3760.99	6710.41 ± 3188.21	5502.06 ± 4209.56	9402.22 ± 4509.39	9881.74 ± 3295.54
	Median	4255.5	6361.76	4977.65	8253.09	10359.35
	Minimum	2686.55	2349.84	1547.45	4728.33	5817.28
	Maximum	12158.57	11004.12	12465.61	16506.18	14328.28
Ash-free Dry Weight (mg/cm^2)	Mean (±SD)	0.31 ± 0.24	0.35 ± 0.11	0.68 ± 0.6	0.33 ± 0.22	0.26 ± 0.17
	Median	0.18	0.35	0.53	0.35	0.18
	Minimum	0.12	0.18	0.18	0.04	0.11
	Maximum	0.7	0.53	1.76	0.53	0.53

Table A3-8: Summary statistics for periphyton in the Lower Columbia River in the summer 2012. Samplers occurred at six locations of increasing relative depth (Shallow to Deep).

Diversity Measures	Statistics	Shallow	Moderate Shallow	Moderate	Moderate Deep	Deep
Abundance (# /$cm^3 \times 10^5$)	Mean (±SD)	9.96 ± 4.59	8.28 ± 5	6.43 ± 1.45	5.59 ± 3.47	6.85 ± 2.61
	Median	10.02	6.58	6.94	5.31	7.4
	Minimum	4.02	3.33	4.3	2.29	2.9
	Maximum	16.48	16.66	7.54	9.44	9.23
Species Richness (#)	Mean (±SD)	37.71 ± 5.74	37.33 ± 2.88	32.75 ± 4.92	32.25 ± 3.3	31.6 ± 4.04
	Median	37	38	32.5	31	29
	Minimum	32	32	27	30	28
	Maximum	48	40	39	37	36
Bio Volume (cm^3/m^2)	Mean (±SD)	2.76 ± 1.4	1.19 ± 0.58	0.95 ± 0.34	1.19 ± 0.8	0.79 ± 0.42
	Median	3.05	0.88	1.01	0.86	0.65
	Minimum	0.83	0.73	0.51	0.67	0.46
	Maximum	4.37	1.94	1.25	2.37	1.53
Simpson's Index	Mean (±SD)	0.81 ± 0.15	0.78 ± 0.1	0.85 ± 0.02	0.85 ± 0.04	0.71 ± 0.18
	Median	0.86	0.8	0.84	0.85	0.78
	Minimum	0.48	0.58	0.83	0.8	0.48
	Maximum	0.88	0.88	0.88	0.9	0.9
Chlorophyll-a ($\mu g/m^2$)	Mean (±SD)	1.47 ± 0.44	1.37 ± 0.55	1.4 ± 0.56	1.39 ± 0.29	1.68 ± 0.42
	Median	1.44	1.28	1.58	1.42	1.59
	Minimum	0.8	0.82	0.61	1.02	1.09
	Maximum	2.09	2.12	1.85	1.71	2.12
Autotrophic Index	Mean (±SD)	3668.04 ± 1398.6	4353.41 ± 1426.61	3976.1 ± 1587.34	5974.89 ± 2563.52	5519.25 ± 1057.55
	Median	3868.64	4090.72	4470.42	6426.23	5802.95
	Minimum	1518.8	2364.17	1719.39	2879.98	4384.45
	Maximum	5931.91	6189.82	5244.15	8167.12	6877.58
Ash-free Dry Weight (mg/cm^2)	Mean (±SD)	0.43 ± 0.14	0.35 ± 0.17	0.35 ± 0	0.26 ± 0.1	0.31 ± 0.09
	Median	0.35	0.35	0.35	0.26	0.35
	Minimum	0.35	0.14	0.35	0.18	0.16
	Maximum	0.7	0.53	0.35	0.35	0.35

Table A3-10: Summary statistics for periphyton in the Lower Columbia River in the winter 2013. Samplers occurred at six locations of increasing relative depth (Shallow to Deep).

Diversity Measures	Statistics	Shallow	Moderate Shallow	Moderate	Moderate Deep	Deep
Abundance (# /$cm^3 \times 10^5$)	Mean (±SD)	16.81 ± 4.66	17.88 ± 7.93	18.44 ± 2.15	24.38 ± 8.63	20.44 ± 2.04
	Median	17.13	17.82	17.84	22.98	19.98
	Minimum	8.42	6.61	15.58	14.77	17.71
	Maximum	23.5	30.45	21.61	38.53	23
Species Richness (#)	Mean (±SD)	36.43 ± 4.28	34.29 ± 2.98	37.71 ± 2.93	37.86 ± 4.3	36.57 ± 3.26
	Median	36	34	37	38	36
	Minimum	31	30	34	33	32
	Maximum	45	39	42	46	43
Bio Volume (cm^3/m^2)	Mean	9.89 ± 5.48	13.38 ± 6.22	14.18 ± 6.38	12.64 ± 6.22	16.25 ± 4.33
	Median	9	14.7	16.21	11.02	15.19
	Minimum	1.51	1.66	5.91	7.24	11.54
	Maximum	16.6	21.39	23.62	23.65	25.09
Simpson's Index	Mean (±SD)	0.91 ± 0.03	0.88 ± 0.08	0.89 ± 0.04	0.85 ± 0.04	0.9 ± 0.04
	Median	0.92	0.9	0.89	0.87	0.9
	Minimum	0.85	0.71	0.83	0.79	0.81
	Maximum	0.93	0.93	0.94	0.91	0.94
Chlorophyll-a (µg/m^2)	Mean (±SD)	6.42 ± 4.18	8 ± 5.33	7.08 ± 3.37	9.78 ± 5.14	9.49 ± 4.87
	Median	8	10.85	5.91	9.45	6.73
	Minimum	1.61	1.65	3.15	5.02	5.89
	Maximum	11.32	14.97	12.58	19.55	18.18
Autotrophic Index	Mean (±SD)	3271.41 ± 2522.33	3023.06 ± 1743.95	2593.83 ± 920.26	3540.31 ± 1187.5	3639.49 ± 915.85
	Median	2432.94	3077.72	3008.94	3112.1	3808.46
	Minimum	1139.1	1031.64	1163.46	2430.08	2428.64
	Maximum	8646.1	6482.12	3576.34	5960.57	5158.18
Ash-free Dry Weight (mg/cm^2)	Mean (±SD)	2.19 ± 1.27	2.82 ± 2.14	3.15 ± 1.75	2.97 ± 1.72	2.72 ± 1.43
	Median	1.41	1.76	3.52	1.76	1.76
	Minimum	0.7	0.88	0.88	1.59	1.41
	Maximum	3.52	7.05	5.29	5.29	5.29

Table A3-11: Summary statistics for periphyton from 7 sampling sites in the Lower Columbia River in the fall 2012. Values are averaged across six locations of increasing relative depth (Shallow to Deep) at each site.

Diversity Measures	Statistics	S1	S2	S3	S4	S5	S6	S7
Abundance (# /$cm^3 \times 10^5$)	Mean (±SD)	8.61 ± 5.84	NA	5.17 ± 1.31	5.92 ± 2.3	5.15 ± 2.51	4.82 ± 3.89	7.43 ± 3.84
	Median	9.73		5.47	6.62	5.48	3.66	6.21
	Minimum	2.3		3.53	2.8	1.89	0.6	2.43
	Maximum	13.81		6.57	8.74	7.75	10.46	11.65
Species Richness (#)	Mean (±SD)	28.67 ± 3.21	NA	28.4 ± 3.21	32.2 ± 5.36	26.5 ± 1.73	27 ± 9.27	30.4 ± 1.52
	Median	30		30	32	27	28	30
	Minimum	25		24	27	24	12	29
	Maximum	31		31	39	28	36	32
Bio Volume (cm^3/m^2)	Mean (±SD)	4.51 ± 3.47	NA	1.3 ± 0.68	2.29 ± 1.44	1.19 ± 0.82	1.42 ± 1.35	2.08 ± 1.04
	Median	5.54		1.48	1.82	0.96	1.2	2.21
	Minimum	0.64		0.56	0.98	0.48	0.1	1.11
	Maximum	7.34		2.22	4.02	2.33	3.55	3.64
Simpson's Index	Mean (±SD)	0.8 ± 0.07	NA	0.83 ± 0.05	0.84 ± 0.03	0.84 ± 0.04	0.85 ± 0.06	0.83 ± 0.03
	Median	0.77		0.86	0.84	0.84	0.85	0.82
	Minimum	0.75		0.74	0.81	0.8	0.75	0.8
	Maximum	0.88		0.86	0.88	0.88	0.91	0.87
Chlorophyll-a ($\mu g/m^2$)	Mean (±SD)	2.47 ± 0.93	NA	2.43 ± 1.04	1.84 ± 0.74	2.45 ± 0.9	1.08 ± 0.38	2.42 ± 0.87
	Median	2.53		1.98	1.5	2.33	1.09	2.5
	Minimum	1.52		1.24	1.23	1.5	0.73	1.27
	Maximum	3.36		3.74	3.08	3.65	1.64	3.65
Autotrophic Index	Mean (±SD)	10349.8 ± 3642.61	NA	5756.24 ± 4746.24	8839.32 ± 2678.56	7658.47 ± 3448.65	6910.05 ± 5783.55	6203.57 ± 2785.48
	Median	9542.64		2686.55	8253.09	6956.38	4642.36	5716.98
	Minimum	7178.47		1865.54	5817.28	4255.5	1547.45	2991.75
	Maximum	14328.28		11262.03	12158.57	12465.61	16506.18	10359.35
Ash-free Dry Weight (mg/cm^2)	Mean (±SD)	0.27 ± 0.14	NA	0.7 ± 0.62	0.24 ± 0.17	0.35 ± 0.14	0.29 ± 0.26	0.46 ± 0.27
	Median	0.35		0.53	0.18	0.35	0.18	0.35
	Minimum	0.11		0.18	0.12	0.18	0.04	0.18
	Maximum	0.35		1.76	0.53	0.53	0.7	0.88

Table A3-12: Summary statistics for periphyton from 7 sampling sites in the Lower Columbia River in the winter 2013. Values are averaged across six locations of increasing relative depth (Shallow to Deep) at each site.

Diversity Measures	Statistics	S1	S2	S3	S4	S5	S6	S7
Abundance (# /$cm^3 \times 10^5$)	Mean (±SD)	20.22 ± 2.12	22.37 ± 5.51	18.11 ± 3.54	26.61 ± 7.81	20.42 ± 4.49	12.96 ± 5.3	16.44 ± 3.82
	Median	19.98	21.87	17.84	23	17.82	14.77	17.61
	Minimum	17.69	17.29	15.16	20	17.13	6.61	10.4
	Maximum	22.98	31.08	23.79	38.53	27.4	19.4	20.74
Species Richness (#)	Mean (±SD)	40.8 ± 4.76	33.4 ± 2.7	36.4 ± 0.89	35.8 ± 1.79	39.8 ± 2.77	33.6 ± 2.3	36.2 ± 2.17
	Median	41	33	36	36	39	34	36
	Minimum	36	30	36	34	36	31	33
	Maximum	46	37	38	38	43	37	39
Bio Volume (cm^3/m^2)	Mean (±SD)	15.1 ± 4.95	14.54 ± 6.08	16.06 ± 5.11	12.6 ± 5.32	16.38 ± 5.28	5.57 ± 4.19	12.61 ± 4.84
	Median	13.52	12.69	17.08	16.12	15.84	5.91	14.77
	Minimum	11.02	9	7.87	6.33	11.05	1.51	5.75
	Maximum	23.62	23.65	21.39	16.71	25.09	11.54	17.41
Simpson's Index	Mean (±SD)	0.91 ± 0.03	0.86 ± 0.05	0.89 ± 0.02	0.85 ± 0.09	0.92 ± 0.02	0.88 ± 0.02	0.89 ± 0.05
	Median	0.92	0.88	0.89	0.9	0.93	0.89	0.9
	Minimum	0.87	0.79	0.87	0.71	0.88	0.85	0.83
	Maximum	0.93	0.9	0.91	0.92	0.94	0.91	0.93
Chlorophyll-a ($\mu g/m^2$)	Mean (±SD)	9.55 ± 2.13	3.62 ± 1.78	8.73 ± 2.05	11.95 ± 3.54	6.57 ± 3.17	4.15 ± 2.37	12.51 ± 6.53
	Median	10.67	3.15	8.58	12.85	6.23	5.3	12.58
	Minimum	5.89	1.82	5.58	6.15	2.56	1.61	4.24
	Maximum	10.91	5.91	11.05	14.97	11.42	6.71	19.55
Autotrophic Index	Mean (±SD)	4991.91 ± 2356.53	2475.07 ± 1120.84	2477.65 ± 580.6	2331.78 ± 766.55	4242.03 ± 1270.93	2588.55 ± 1066.57	3388.35 ± 1172.75
	Median	4180.28	2845.6	2432.94	2430.08	3817.05	3008.94	3438.79
	Minimum	3077.72	1031.64	1581.84	1163.46	3352.82	1139.1	2269.6
	Maximum	8646.1	3576.34	3133.6	3210.97	6482.12	3808.46	5158.18
Ash-free Dry Weight (mg/cm^2)	Mean (±SD)	2.26 ± 1.17	1.52 ± 0.39	3.52 ± 0	5.29 ± 1.25	1.52 ± 0.46	1.52 ± 0.39	3.77 ± 1.67
	Median	1.59	1.76	3.52	5.29	1.76	1.76	3.52
	Minimum	1.23	0.88	3.52	3.52	0.7	0.88	1.23
	Maximum	3.52	1.76	3.52	7.05	1.76	1.76	5.29

Appendix A4. Benthic Invertebrate Summary Tables (2008 to 2013).

Table A4-1: Relative abundance and relative biomass of benthic invertebrates in the fall of each year from 2008 to 2012.

2008

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Hydropsyche	22.21	Trichoptera	56.32
Parachironomus	17.01	Gastropoda	18.42
Tvetenia discoloripes gr.	12.82	Acari.Crustacea	16.86
Orthocladius Complex	8.87	Chironomidae	4.66
Cheumatopsyche	4.72	Ephemeroptera	2.01
Synorthocladius	4.35	Diptera	1.44
Cricotopus bicinctus gr.	3.99	Bivalvia	0.14
Ephemerella	3.00	Plecoptera	0.13
Orthocladiinae	2.93	Oligochaeta (+ Annelids 2008-2010)	0.02
Simulium	2.54		

2009

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Simulium	24.70	Acari.Crustacea	37.21
Hydropsyche	19.35	Trichoptera	35.37
Parachironomus	13.28	Gastropoda	13.22
Tvetenia discoloripes gr.	5.74	Diptera	9.97
Synorthocladius	5.47	Chironomidae	2.36
Cricotopus bicinctus gr.	4.35	Ephemeroptera	0.95
Nais behningi	2.86	Plecoptera	0.73
Baetis	2.43	Hirudinea (+ Large Annelids 2008-2010)	0.18
Ceraclea	2.21		
Ephemerellidae	2.12		

2010

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Parachironomus	16.69	Acari.Crustacea	36.97
Hydropsyche	12.94	Trichoptera	29.19
Ephemerellidae	9.10	Gastropoda	26.89
Orthocladius Complex	8.73	Diptera	3.16
Simulium	7.43	Ephemeroptera	2.06
Simuliidae	7.01	Chironomidae	1.66
Rheotanytarsus	6.08	Plecoptera	0.04
Synorthocladius	4.17	Bivalvia	0.03
Tvetenia discoloripes gr.	3.49		
Nais behningi	2.57		

2012

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Tvetenia discoloripes gr.	17.91	Trichoptera	54.47
Hydropsyche	13.92	Gastropoda	29.94
Hydropsychidae	8.59	Odonata	4.73
Parachironomus	8.58	Acari.Crustacea	4.35
Ephemerellidae	8.01	Diptera	2.07
Orthocladius Complex	6.84	Oligochaeta (+ Annelids 2008-2010)	1.31
Simulium	6.51	Hirudinea (+ Large Annelids 2008-2010)	1.16
Eukiefferiella claripennis gr.	4.40	Chironomidae	0.99
Pagastia	2.60	Ephemeroptera	0.90
Cheumatopsyche	2.20	Bivalvia	0.04

Table A4-2: Relative abundance and relative biomass of benthic invertebrates in the summer of each year from 2008 to 2012.

2008

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Hydropsyche	19.92	Trichoptera	38.13
Cheumatopsyche	13.07	Acari Crustacea	31.38
Hydropsychidae	11.40	Gastropoda	19.19
Tvetenia discoloripes gr.	8.45	Oligochaeta (+ Annelids 2008-2010)	4.64
Synorthocladius	4.95	Diptera	3.10
Cricotopus bicinctus gr.	3.79	Chironomidae	2.82
Parachironomus	3.40	Ephemeroptera	0.68
Ceraclea	3.33	Coleoptera	0.05
Crangonyx	2.60	Other	0.02
Gastropoda	2.44	Bivalvia	0.00

2009

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Hydropsychidae	20.13	Acari Crustacea	37.88
Synorthocladius	13.23	Gastropoda	22.32
Simulium	7.78	Diptera	11.58
Parachironomus	7.06	Chironomidae	8.99
Cricotopus bicinctus gr.	5.56	Trichoptera	8.11
Lymnaeidae	4.04	Ephemeroptera	6.29
Tvetenia discoloripes gr.	4.00	Hirudinea (+ Large Annelids 2008-2010)	4.38
Nais	2.62	Bivalvia	0.43
Crangonyx	2.01	Coleoptera	0.03
Hydropsyche	1.94	Odonata	0.00

2010

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Hydropsychidae	21.99	Acari Crustacea	33.62
Synorthocladius	6.99	Trichoptera	31.02
Rheotanytarsus	6.31	Gastropoda	15.67
Lymnaeidae	5.60	Chironomidae	8.06
Trichoptera	5.34	Diptera	6.32
Simulium	4.88	Ephemeroptera	3.64
Planorbidae	4.04	Hirudinea (+ Large Annelids 2008-2010)	1.66
Nais behningi	3.04	Bivalvia	0.02
Nais	3.02	Coleoptera	0.00
Parachironomus	2.79	Odonata	0.00

2012

Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Hydropsychidae	43.19	Trichoptera	50.74
Hydropsyche	6.17	Gastropoda	18.64
Brachycentrus occidentalis	5.95	Hirudinea (+ Large Annelids 2008-2010)	10.92
Tvetenia discoloripes gr.	4.87	Diptera	6.80
Simuliidae	4.85	Ephemeroptera	5.34
Simulium	4.48	Chironomidae	4.21
Caudatella	4.07	Odonata	1.41
Parachironomus	3.20	Plecoptera	1.28
Orthocladius Complex	2.86	Oligochaeta (+ Annelids 2008-2010)	0.32
Synorthocladius	1.82	Acari.Crustacea	0.25

Table A4-3: Relative abundance and relative biomass of benthic invertebrates in the winter of 2013.

2013			
Species	Relative Abundance (%)	Class / Order	Relative Biomass (%)
Simulium	26.08	Gastropoda	25.83
Synorthocladius	16.24	Diptera	23.43
Orthocladius Complex	14.93	Trichoptera	23.09
Simuliidae	9.46	Ephemeroptera	11.64
Ephemerellidae	4.57	Chironomidae	9.19
Ephemerella	4.16	Plecoptera	4.82
Tvetenia discoloripes gr.	3.98	Acari.Crustacea	0.77
Tanytarsus	3.03	Oligochaeta (+ Annelids 2008-2010)	0.64
Diamesa	2.66	Bivalvia	0.31
Hydropsyche	1.96	Hirudinea (+ Large Annelids 2008-2010)	0.28

Table A4-4: Summary statistics for benthic invertebrates in the Lower Columbia River in the fall from 2008-2012. Samplers occurred at six locations of increasing relative depth (Shallow to Deep).

Diversity Measure	Statistics	Shallow	Moderate Shallow	Moderate	Moderate Deep	Deep
Abundance (#/basket)	Mean (±SD)	2062.36 ± 2133.91	4347.67 ± 6670.38	3850.62 ± 5626.03	13772.33 ± 9926.35	4265.31 ± 4929.13
	Median	1142	739	1404	17865	1594.87
	Minimum	31	206	76	2454	363.02
	Maximum	8525	16999	20800	20998	14133
Biomass (g/basket)	Mean (±SD)	2.37 ± 5.89	1.19 ± 1.55	1.19 ± 1.3	4.22 ± 4.61	2.05 ± 3.7
	Median	0.8	0.78	0.7	3.43	0.97
	Minimum	0.02	0.09	0.01	0.06	0.11
	Maximum	30.25	4.27	5.22	9.17	16.4
Species Richness (#)	Mean (±SD)	24.58 ± 6.6	21.5 ± 4.85	22.22 ± 7.72	21 ± 5.57	23.44 ± 7.85
	Median	25	20.5	20	20	21.5
	Minimum	8	15	10	16	14
	Maximum	36	28	42	27	41
EPT Richness (#)	Mean (±SD)	6.12 ± 2.12	7 ± 2.61	6.83 ± 2.72	5.33 ± 0.58	7.17 ± 2.41
	Median	6	7.5	7	5	6.5
	Minimum	3	3	2	5	4
	Maximum	11	10	13	6	11
Percent EPT (%)	Mean (±SD)	0.28 ± 0.21	0.24 ± 0.13	0.35 ± 0.2	0.38 ± 0.24	0.37 ± 0.2
	Median	0.3	0.23	0.38	0.44	0.35
	Minimum	0.03	0.08	0.09	0.12	0.1
	Maximum	0.84	0.42	0.85	0.58	0.76
Percent Chironomidae (%)	Mean (±SD)	0.48 ± 0.22	0.51 ± 0.1	0.47 ± 0.25	0.58 ± 0.2	0.46 ± 0.24
	Median	0.52	0.52	0.51	0.54	0.45
	Minimum	0.1	0.37	0.11	0.41	0.09
	Maximum	0.88	0.62	0.89	0.8	0.86
Simpson's Index	Mean (±SD)	0.67 ± 0.29	0.12 ± 0.03	0.58 ± 0.29	0.24 ± 0.07	0.58 ± 0.27
	Median	0.82	0.12	0.72	0.22	0.7
	Minimum	0.08	0.08	0.1	0.18	0.12
	Maximum	0.9	0.17	0.9	0.31	0.88
Hilsenhoff Biotic Index	Mean (±SD)	5.53 ± 1.05	5.24 ± 0.31	5.56 ± 1.11	4.88 ± 0.8	5.42 ± 1.41
	Median	5.22	5.27	5.21	4.62	5.22
	Minimum	3.79	4.8	4.32	4.24	3.7
	Maximum	7.92	5.72	8.93	5.78	8.78

Table A4-5: Summary statistics for benthic invertebrates in the Lower Columbia River in the summer from 2008-2012. Samplers occurred at six locations of increasing relative depth (Shallow to Deep).

Diversity Measure	Statistics	Shallow	Moderate Shallow	Moderate	Moderate Deep	Deep
Abundance (#/basket)	Mean (±SD)	4751.89 ± 5903.14	4954.83 ± 5974.24	2635.01 ± 2979.4	1053 ± 693.8	4299.88 ± 6289.58
	Median	2391.47	1878	1548.73	883	2215.91
	Minimum	84	388	65	460	77
	Maximum	25886.08	14300	13946.36	1816	28924.05
Biomass (g/basket)	Mean (±SD)	1.4 ± 1.61	0.49 ± 0.46	0.74 ± 0.69	0.32 ± 0.18	0.67 ± 0.64
	Median	0.86	0.43	0.57	0.29	0.5
	Minimum	0.16	0.06	0.03	0.16	0.03
	Maximum	6.83	1.22	3.21	0.52	2.65
Species Richness (#)	Mean (±SD)	25.48 ± 8.84	25.67 ± 9.42	24.94 ± 9.4	32.33 ± 13.43	21.82 ± 9.56
	Median	24	24	25	38	21
	Minimum	12	16	10	17	7
	Maximum	48	42	51	42	49
EPT Richness (#)	Mean (±SD)	4.78 ± 1.97	6.5 ± 1.87	5.14 ± 1.68	6 ± 2.65	4.97 ± 1.78
	Median	4.5	6	5	7	5
	Minimum	2	5	1	3	1
	Maximum	11	10	8	8	10
Percent EPT (%)	Mean (±SD)	0.36 ± 0.28	0.61 ± 0.25	0.4 ± 0.26	0.41 ± 0.25	0.48 ± 0.25
	Median	0.32	0.69	0.4	0.38	0.51
	Minimum	0.01	0.2	0.01	0.18	0.01
	Maximum	0.93	0.85	0.87	0.68	0.86
Percent Chironomidae (%)	Mean (±SD)	0.35 ± 0.18	0.18 ± 0.1	0.28 ± 0.16	0.2 ± 0.14	0.29 ± 0.16
	Median	0.33	0.18	0.3	0.2	0.26
	Minimum	0.02	0.05	0.02	0.07	0.04
	Maximum	0.82	0.32	0.68	0.34	0.7
Simpson's Index	Mean (±SD)	0.73 ± 0.24	0.35 ± 0.19	0.74 ± 0.19	0.15 ± 0.08	0.72 ± 0.21
	Median	0.82	0.36	0.8	0.12	0.76
	Minimum	0.08	0.1	0.24	0.09	0.12
	Maximum	0.94	0.55	0.94	0.24	0.96
Hilsenhoff Biotic Index	Mean (±SD)	5.31 ± 1	4.22 ± 0.49	5.15 ± 0.9	4.07 ± 0.6	5.12 ± 0.9
	Median	5.16	4.15	4.97	4.24	5.03
	Minimum	3.34	3.67	3.31	3.4	3
	Maximum	7.51	5.08	7.44	4.56	7.32

Table A4-6: Summary statistics for benthic invertebrates in the Lower Columbia River in the winter of 2013. Samplers occurred at six locations of increasing relative depth (Shallow to Deep).

Diversity Measure	Statistics	Shallow	Moderate Shallow	Moderate	Moderate Deep	Deep
Abundance (#/basket)	Mean (±SD)	2262.43 ± 2623.28	2564.43 ± 3233.84	4375.62 ± 5455.81	3562.29 ± 3622.34	7080.86 ± 10443.93
	Median	808	1523	1755.5	2891	1802
	Minimum	221	600	155	249	228
	Maximum	6440	9780	13060	10100	29460
Biomass (g/basket)	Mean (±SD)	0.56 ± 0.65	0.55 ± 0.6	1.94 ± 2.85	0.42 ± 0.31	0.82 ± 1.25
	Median	0.36	0.27	0.46	0.43	0.19
	Minimum	0.11	0.08	0.08	0.03	0.08
	Maximum	1.95	1.51	6.54	0.86	3.43
Species Richness (#)	Mean (±SD)	19 ± 6.51	19.29 ± 4.07	18.88 ± 3.14	17.57 ± 4.65	17.00 ± 5.54
	Median	19	19	20	17	15
	Minimum	9	14	14	10	11
	Maximum	29	26	22	24	24
EPT Richness (#)	Mean (±SD)	5.14 ± 1.77	6.71 ± 0.95	7.75 ± 2.38	7.57 ± 3.05	6.29 ± 2.63
	Median	5	7	7	7	8
	Minimum	2	5	5	3	2
	Maximum	7	8	11	13	9
Percent EPT (%)	Mean (±SD)	0.09 ± 0.05	0.15 ± 0.06	0.18 ± 0.14	0.14 ± 0.11	0.13 ± 0.11
	Median	0.1	0.12	0.16	0.09	0.05
	Minimum	0.01	0.07	0.02	0.01	0.01
	Maximum	0.15	0.25	0.48	0.33	0.29
Percent Chironomidae (%)	Mean (±SD)	0.57 ± 0.2	0.59 ± 0.2	0.47 ± 0.26	0.39 ± 0.27	0.36 ± 0.28
	Median	0.61	0.64	0.5	0.31	0.43
	Minimum	0.25	0.15	0.09	0.08	0.03
	Maximum	0.79	0.78	0.83	0.72	0.71
Simpson's Index	Mean (±SD)	0.21 ± 0.11	0.18 ± 0.07	0.23 ± 0.13	0.28 ± 0.16	0.33 ± 0.26
	Median	0.18	0.15	0.2	0.23	0.19
	Minimum	0.07	0.11	0.11	0.11	0.09
	Maximum	0.4	0.31	0.54	0.55	0.82
Hilsenhoff Biotic Index	Mean (±SD)	4.64 ± 1.09	4.14 ± 0.75	4.08 ± 0.46	4.24 ± 0.37	3.9 ± 0.92
	Median	4.57	3.98	4.05	4.16	3.94
	Minimum	3.42	3.31	3.42	3.67	2.89
	Maximum	6.52	5.38	4.79	4.63	5.58

Table A4-7: Summary statistics for benthic invertebrates in the Lower Columbia River in the fall from 2008-2012.

Diversity Measure	Statistics	2008	2009	2010	2012
Abundance (# /basket)	Mean (±SD)	2014.24 ± 1792.41	1508.66 ± 1601.04	1801.3 ± 1075.45	7550.77 ± 7290.58
	Median	1203.21	1361.26	1402	5124
	Minimum	31	130	238	76
	Maximum	5936.4	6638.3	3721.49	20998
Biomass (g/basket)	Mean (±SD)	3.27 ± 6.8	0.92 ± 0.54	0.99 ± 0.85	2.18 ± 3.52
	Median	0.94	0.96	0.84	0.89
	Minimum	0.02	0.14	0.22	0.01
	Maximum	30.25S	2.1	3.75	16.4
Species Richness (#)	Mean (±SD)	23.84 ± 8.3	18.82 ± 5.16	27.38 ± 7.59	21.73 ± 5.7
	Median	25	17.5	25	21
	Minimum	8	11	19	11
	Maximum	36	33	42	33
EPT Richness (#)	Mean (±SD)	5.74 ± 2.51	5.45 ± 1.5	7 ± 2.22	7.58 ± 2.44
	Median	5	5.5	7	7
	Minimum	2	3	3	3
	Maximum	11	8	11	13
Percent EPT (%)	Mean (±SD)	0.34 ± 0.23	0.34 ± 0.19	0.3 ± 0.22	0.34 ± 0.17
	Median	0.35	0.36	0.26	0.35
	Minimum	0.08	0.03	0.03	0.08
	Maximum	0.85	0.65	0.76	0.69
Percent Chironomidae (%)	Mean (±SD)	0.58 ± 0.24	0.31 ± 0.22	0.44 ± 0.25	0.5 ± 0.15
	Median	0.58	0.2	0.45	0.5
	Minimum	0.1	0.09	0.12	0.12
	Maximum	0.89	0.74	0.8	0.8
Simpson's Index	Mean (±SD)	0.75 ± 0.15	0.74 ± 0.11	0.81 ± 0.11	0.17 ± 0.07
	Median	0.78	0.74	0.84	0.16
	Minimum	0.42	0.41	0.54	0.08
	Maximum	0.9	0.88	0.9	0.31
Hilsenhoff Biotic Index	Mean (±SD)	5.99 ± 1.22	5.42 ± 0.92	5.7 ± 1.21	4.81 ± 0.66
	Median	5.54	5.04	5.28	4.72
	Minimum	4.88	4.45	3.97	3.7
	Maximum	8.93	7.92	7.98	6.54

Table A4-8: Summary statistics for benthic invertebrates in the Lower Columbia River in the summer from 2008-2012.

Diversity Measure	Statistics	2008	2009	2010	2012
Abundance (# /basket)	Mean (±SD)	3588.76 ± 4338.59	2733.98 ± 5222.22	4901.58 ± 6361.52	4507.38 ± 4831.92
	Median	1620.82	1389.88	2792	2577.5
	Minimum	84	249	65	208
	Maximum	13946.36	28924.05	25886.08	16840
Biomass (g/basket)	Mean (±SD)	1.2 ± 1.29	0.7 ± 0.56	1.02 ± 1.41	0.71 ± 0.93
	Median	0.82	0.59	0.76	0.41
	Minimum	0.18	0.03	0.07	0.06
	Maximum	6.83	2.11	6.69	4.16
Species Richness (#)	Mean (±SD)	21.22 ± 7.51	26.97 ± 9.75	24.94 ± 10.34	25 ± 9.31
	Median	20.5	25	23	24
	Minimum	7	11	12	13
	Maximum	36	49	51	42
EPT Richness (#)	Mean (±SD)	5.09 ± 1.87	4.65 ± 1.91	4.94 ± 1.79	5.71 ± 1.76
	Median	5	4	5	6
	Minimum	1	2	1	3
	Maximum	11	10	8	10
Percent EPT (%)	Mean (±SD)	0.49 ± 0.27	0.27 ± 0.22	0.36 ± 0.25	0.59 ± 0.23
	Median	0.58	0.23	0.36	0.64
	Minimum	0.01	0.01	0.01	0.11
	Maximum	0.87	0.77	0.89	0.93
Percent Chironomidae (%)	Mean (±SD)	0.28 ± 0.15	0.41 ± 0.18	0.29 ± 0.16	0.19 ± 0.1
	Median	0.28	0.41	0.3	0.18
	Minimum	0.02	0.13	0.02	0.05
	Maximum	0.67	0.82	0.59	0.41
Simpson's Index	Mean (±SD)	0.78 ± 0.11	0.8 ± 0.12	0.79 ± 0.15	0.31 ± 0.19
	Median	0.8	0.85	0.82	0.25
	Minimum	0.54	0.52	0.2	0.08
	Maximum	0.93	0.94	0.96	0.79
Hilsenhoff Biotic Index	Mean (±SD)	5.48 ± 0.83	5.38 ± 0.86	5.33 ± 0.77	4.03 ± 0.56
	Median	5.1	5.15	5.12	4.01
	Minimum	4.58	4.31	4.34	3
	Maximum	7.51	7.33	7.44	5.4

Table A4-9: Summary statistics for benthic invertebrates in the Lower Columbia River in the winter of 2013.

Diversity Measure	Statistics	2013
Abundance (# /basket)	Mean (±SD)	3980.42 ± 5732.21
	Median	1704
	Minimum	155
	Maximum	29460
Biomass (g/basket)	Mean (±SD)	0.89 ± 1.54
	Median	0.36
	Minimum	0.03
	Maximum	6.54
Species Richness (#)	Mean (±SD)	18.36 ± 4.67
	Median	19
	Minimum	9
	Maximum	29
EPT Richness (#)	Mean (±SD)	6.72 ± 2.35
	Median	7
	Minimum	2
	Maximum	13
Percent EPT (%)	Mean (±SD)	0.14 ± 0.1
	Median	0.12
	Minimum	0.01
	Maximum	0.48
Percent Chironomidae (%)	Mean (±SD)	0.48 ± 0.25
	Median	0.56
	Minimum	0.03
	Maximum	0.83
Simpson's Index	Mean (±SD)	0.24 ± 0.16
	Median	0.19
	Minimum	0.07
	Maximum	0.82
Hilsenhoff Biotic Index	Mean (±SD)	4.2 ± 0.76
	Median	4.09
	Minimum	2.89
	Maximum	6.52