Columbia River Project Water Use Plan

LOWER COLUMBIA RIVER FISH

Reference: CLBMON-44

Lower Columbia River Physical Habitat and Ecological Productivity Monitoring

Study Period: 2008 – 2019

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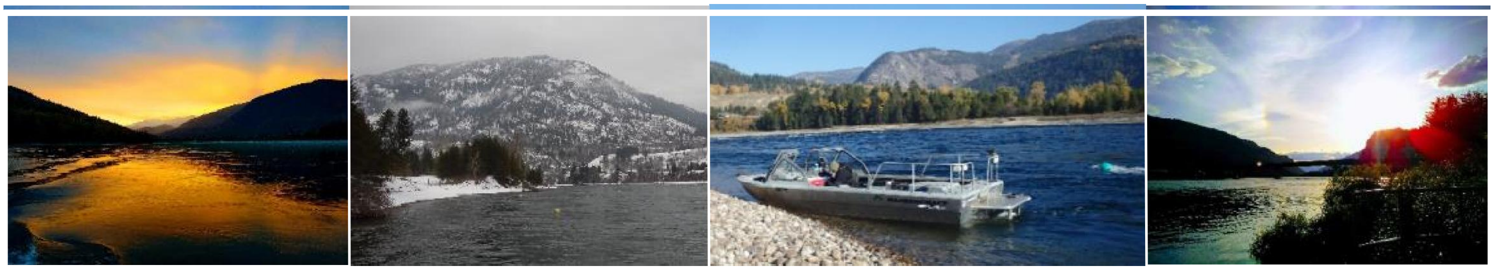
August 31, 2019

Lower Columbia River Fish

Monitoring Program No. CLBMON-44

Lower Columbia River Physical Habitat and Ecological Productivity Monitoring



Final Report

Prepared for

BC Hydro Generation
Water Licence Requirements

6911 Southpoint Drive
Burnaby, BC

BC Hydro Reference # EC11-421945

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Cover photos:

From left to right: 2017 Genelle; 2016 winter deployment (S7); 2018 fall retrieval (S4); 2018 spring deployment (S2)

EXECUTIVE SUMMARY

This is the final report of a twelve-year study (2008 – 2019) 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 in southern British Columbia. Discharges from the HLK Dam during winter and spring have the potential to affect salmonid spawning and rearing habitats. To minimize impacts, BC Hydro and Power Authority (BC Hydro) altered operations of HLK Dam to include: 1) Rainbow Trout protection flows that stabilize HLK discharges from April 1 through June 30, to reduce redd dewatering and subsequent egg loss of rainbow trout, and 2) Mountain Whitefish flows that limit maximum discharges during peak spawning in January and later stabilizes discharges to reduce egg dewatering until Mountain Whitefish larvae emerge in late March.

The objective of this study was to examine the influence of the managed flow periods (Mountain Whitefish (MWF) Jan 1 - Mar 31; and Rainbow Trout (RBT) Apr 1 - Jun 30; and fall fluctuating flow (FFF) Sep 1 - Oct 31) on select physical habitat and ecological productivity measures. Benthic productivity, inclusive of periphyton and macrobenthic invertebrates, are key factors in a river system because they are a primary food source for fish.

The benthic productivity of LCR was investigated using artificial Styrofoam samplers for periphyton and rock baskets for benthic invertebrates. Samplers were placed along transects at 7 sites in Reach 2 during three different seasons (winter, summer and fall) for a duration between 6 and 12 weeks to allow for the growth and establishment of periphyton and benthic invertebrates. Physical river parameters including water quality, elevation, temperature, substrate size, and velocity were also measured.

A variety of statistical tests were used to determine if managed MWF, RBT and FFF resulted in changes in community composition and an increase of total periphyton and benthic invertebrate biomass accruals. Study results indicate that managed flow periods affected water quality and benthic productivity by stabilizing flows to reduce peaks and velocities, as well as increase substrate exposure during low flows. For water quality, this meant stabilized electrochemistry, lower turbidity and more consistent nutrient delivery to the periphyton. In turn, managed flows and water chemistry encouraged more periphyton growth particularly of the large or filamentous types. All managed flow periods are expected to induce larger periphyton biomass. Similarly, the benthic invertebrate communities benefitted from managed RBT and FFF flows because they maintain a greater area of wetted habitat and higher velocities, both of which are expected to increase the availability of fish food organisms, while lower winter flows with managed MWF flows are expected to decrease the availability of fish food due to smaller areas of wetted habitat.

Our findings concur with the literature, which clearly demonstrates that conditions including flow, velocity, water chemistry and substrates play interconnected roles in the overall characterization and growth rates of riverine benthic communities. The scale of the differences made by managed flows can be overshadowed by extreme flow events such as a large freshet during RBT flows or a severe storm in FFF. When flows are stable, the managed flow periods play a significant role in shaping LCR benthic communities and their productivity.

ACRONYMS AND ABBREVIATIONS

AFDW	ash free dry weight
ALR	Arrow Lakes Reservoir
BBK	Birchbank
BC Hydro	British Columbia Hydro and Power Authority
BRD	Combined discharge from Brilliant Dam, including spill and the Brilliant Dam expansion project
CART	Classification and Regression Tree
Caro Labs	Caro Environmental Laboratories (Kelowna, B.C.)
Celgar	Zellstoff Celgar Mill
CFU	colony forming unit
chl-a	chlorophyll-a
CRIEMP	Columbia River Integrated Environmental Monitoring Program
Didymo	*Didymosphenia geminate*
DIN	dissolved inorganic nitrogen
DO	dissolved oxygen
EPT	Ephemeroptera (mayflies), Plecoptera (stoneflies), Trichoptera (caddisflies)
FFF	fall fluctuating flow
HBI	Hilsenhoff Biotic Index
HLK	Hugh L. Keenleyside
GAM	Generalized Additive Model
QA/QC	quality assurance, quality control
km	kilometer
L	litre
LCR	Lower Columbia River
m	metre
m asl	metres above sea level
max	maximum value
MCR	Middle Columbia River
min	minimum value
MWF	Mountain Whitefish
n	sample size
N	nitrogen
NMDS	non-metric multidimensional scaling
NTU	nephelometric turbidity units
PERMANOVA	permutational multivariate analysis of variance
POM	particulate organic material
RBT	Rainbow Trout
SD	standard deviation
STD	standardized
T-P	total phosphorus
TDS	total dissolved solids
TSS	total suspended solids
WQIS	water quality index station
UTM	Universal Transverse Mercator
WUP CC	Columbia River Water Use Plan Consultative Committee

DEFINITIONS

Term	Definition
Accrual rate	A function of cell settlement, actual growth and losses (grazing, sloughing)
Autotrophic	An organism capable of synthesizing its own food from inorganic substances, using light or chemical energy
Benthic	Organisms that dwell in or are associated with the sediments
Benthic production	The production within the benthos originating from both periphyton and benthic invertebrates
Catastrophic flow	Flow events that have population level consequences of >50% mortality
Cyanobacteria	Bacteria-like algae having cyanochrome as the main photosynthetic pigment
Diatoms	Algae that have hard, silica-based "shells" frustules
Diel	Denoting or involving a period of 24 hours
Epilithic algae	Algae that grow on hard inert substrates, such as gravel, cobbles, boulders
Eutrophic	Nutrient-rich, biologically productive water body
Flow	The instantaneous volume of water flowing at any given time (e.g.1200 m^3/s)
Freshet	The flood of a river from melted snow in the spring
Functional Feeding group	(FFG) Benthic invertebrates can be classified by mechanism by which they forage, referred to as functional feeding or foraging groups
Guilds of algae	Low-profile guild includes small tightly adhering taxa; motile guild can move; planktonic guild is from lakes; high profile guild are large, exposed to flows (stalked, filamentous etc.)
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.
Light attenuation	Reduction of sunlight strength during transmission through water
Limitation, nutrient	A nutrient can limit or control the potential growth of organisms e.g. P or N
Linear Regression Model	Linear regression attempts to model the relationship between two variables by fitting a linear equation to observed data
Macroinvertebrate	An invertebrate that is large enough to be seen without a microscope
Mainstem	The primary downstream segment of a river, as contrasted to its tributaries
Mesotrophic	A body of water with moderate nutrient concentrations
Microflora	The sum of algae, bacteria, fungi, *Actinomycetes,* etc., in water or biofilms
Morphology, river	The study of channel pattern and geometry at several points along a river
Peak biomass	The highest density, biovolume or chl-a attained in a set time on a substrate
Periphyton	Microflora that are attached to aquatic plants or solid substrates
Phytoplankton	Algae that float, drift or swim in water columns of reservoirs and lakes
Ramping of flows	A progressive change of discharge into a stream or river channel
Redd	A spawning nest made by a fish, especially a salmon or trout
Riffle	A stretch of choppy water in a river caused by a shoal or sandbar
Riparian	The interface between land and a stream or lake
Salmonid	Pertaining to the family *Salmonidae*, including the salmons, trouts, chars, and whitefishes.
Substrates	Substrate (sediment) is the material (boulder cobble sand silt clay) on the stream bottom
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

Management Question	**Summary of Key Monitoring Results**
Physical Habitat Monitoring MQ-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?	The influence of flow on LCR water temperature was relatively small compared to other predictors such as reservoir elevation, reservoir temperature and air temperature at sites upstream of the Kootenay River. At downstream sites, flow was a much more important predictor of water temperature, but it was largely due to the influence of Kootenay River flows which are slightly warmer than LCR flows, and not due to the managed MWF, RBT or FFF regimes from HLK Dam. Given the small influence of HLK flows on LCR water temperature and the lack of evidence that MWF, RBT or FFF regimes affect LCR water temperature, we **accept the null hypothesis** that that continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not alter the seasonal water temperatures regime of LCR.
Physical Habitat Monitoring MQ-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?	Analyses indicate that river flow is an important determinant of water levels. At all locations, the river level difference between MWF maximum peak spawning and minimum incubation was greater during pre-MWF flows than during post MWF flows. Similarly, river elevation data from monitoring stations WQIS2 and WQIS3 were regressed with flow data. The cumulative elevation drops that occurred during pre-RBT flows (1984-1991) were significantly higher than those during post (1992-2007) and continuous (2008-2018) flow periods. We therefore **reject all three** (Ho_{2phy}, Ho_{2Aphy}, Ho_{2Bphy}) null hypotheses: *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.*

Physical Habitat Monitoring MQ-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?	We **reject the management hypothesis Ho_{3Aphy}**, that states that managed MWF flows have no effect on the water quality of LCR. The lines of evidence to support this rejection of hypothesis Ho_{3Aphy} include: • The comparison of 2013 MWF flows (similar hydrograph to unmanaged flows) with 2014 and 2015 showed elevated conductivity and nitrate concentrations in 2013. • Descriptive statistics suggested that MWF managed flow periods may influence total phosphorus concentrations. • Operations such as Celgar and sewage outflows can increase the range of T-P values observed during winter low flows (low dilution), and this should be evident in the first half of the MWF flow period. • T-N results indicate that there are additional nitrogen sources in the LCR that augment concentrations in the flows from ALR above its confluence with Kootenay River. These sources likely experience less dilution with managed MWF flows than unmanaged flows over the same period. We **reject the management hypothesis Ho_{3Bphy}**, stating that managed RBT flows have no effect on the water quality of LCR. The lines of evidence to support this rejection of hypothesis Ho_{3Bphy} include: • Flow variability increased the availability of some nutrients (NO_2+NO_3 and total P), and RBT flows restrict flow variability. • RBT managed flow periods may influence total phosphorus concentrations. • The comparison of 2012 RBT (extreme freshet) to the other RBT flow years showed significantly lower conductivity, higher nitrate and lower T-P concentrations compared to other RBT flow periods. • The stabilized RBT flows should lower organic and total nitrogen concentrations by reducing scour. • Turbidity and TSS are positively correlated with flows, so lower peak velocities in RBT flow periods would limit turbid conditions. We **reject the management hypothesis Ho_{3Cphy}**, that states that managed FFF flows have no effect on the water quality of LCR. The lines of evidence to support this rejection of hypothesis Ho_{3Cphy} include: • Substrate exposure occurs during very low flows and can affect rates of groundwater influx and organic decay, particularly at shallow and mid shallow sites and substrate exposure was reduced under the first half of the fall fluctuating flow period. • The higher averaged flows in the first half of the FFF period should decrease electrochemistry concentrations through dilution but increase organic and total nutrient concentrations through scour.

Ecological Productivity Monitoring MQ-1 What is the composition, abundance, and biomass of epilithic algae in LCR? 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 epilithic algae?	***We reject the management hypothesis*** *Ho_{1eco} stating that 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.* Although river discharge (flow variability and velocity) clearly influenced the LCR periphyton community, the influences of managed fish flows (MWF, RBT and FFF) are more difficult to discern. We used the following lines of evidence: ***We reject Ho_{2Aeco}:*** *Continued implementation of MWF does not increase total biomass accrual of periphyton in LCR.* • Comparison of winter 2013 (flows were similar to unregulated flows) productivity metrics with winter 2014, 2016 and 2018 showed higher chl-a in 2016 compared to 2013, but lower biovolume in all years (and significantly so in 2014) compared to reference year 2013 with a flow regime similar to unmanaged winter flows. • In this study, lower stable flows during winter with MWF managed flows were associated with higher biomass, particularly of the high profile guild, and slower growth rates. • The transect depth where peak biomass occurred was MS to M in winter but shifted to MD in fall, indicating that flow regime affects periphyton productivity. ***We reject Ho_{2Beco}:*** *Continued implementation of RBT flows does not increase total biomass accrual of periphyton in LCR.* • Sites with higher velocities had lower periphyton production, and the unmanaged hydrograph had higher peak freshet flows, like reference year 2012, suggesting that unmanaged flows would decrease periphyton productivity. • During the summer RBT flow period, peak velocities and flow variability are reduced and percent high profile guild density significantly higher in years with moderate RBT flows compared to the extreme flows in 2012. • With management, substrate exposure to air was reduced, increasing the surface area available for periphyton productivity in shallow sites ***We reject Ho_{2Ceco}:*** *Continued fluctuations of flow during the fall do not increase total biomass accrual of periphyton in LCR.* • Fall periphyton production data indicate that FFF flow variability decreased total biomass accrual. Thus, decreased flow variability with managed FFF flows compared to unmanaged fall flows should allow greater periphyton biovolume and chl-a. • Substrate dewatering was greatest in FFF and resulted in loss of periphyton productivity in the shallows. Substrate dewatering was less severe and sustained with FFF managed flows than it was under unmanaged fall flows and should increase periphyton productivity.

Management Question	**Summary of Key Monitoring Results**
Ecological Productivity Monitoring MQ-2 What is the composition, abundance, and biomass of benthic invertebrates in LCR? 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?	Managed MWF flows appear to result in less submerged habitat, **we reject the management hypothesis** that MWF flows during winter, does not affect the biomass, abundance and composition of benthic invertebrates in LCR. We believe that these metrics are affected by a reduction in overall wetted habitat. By stabilizing freshet flows and eliminating the highest and lowest flows, the continued implementation of RBT flows results in a more stable benthic invertebrate community that utilizes the greater availability of wetted substrate. **We therefore reject the null hypothesis** that the continued implementation of RBT flows does not affect the biomass, abundance or composition of benthic invertebrates in LCR. The overall higher and stable flows during the FFF period, also results in a more stable benthic invertebrate community due to a greater area of submerged habitat. Prolonged dewatering results in losses to invertebrate abundance, biomass and diversity. Based on this, **we reject the hypothesis** that the continued implementation of FFF does not affect the biomass, abundance or composition of benthic invertebrates in LCR.
Ecological Productivity Monitoring MQ-3 Are organisms that are used as food by juvenile and adult MWF and RBT in LCR supported by benthic production in LCR?	Since the managed MWF, RBT and FFF periods have resulted in changes to the area of submerged habitat, as well as changes to river velocities, **we accept the management hypothesis** $\mathbf{HO_{3Aeco}}$, that MWF flows does not increase the availability of fish food organisms but **reject** $\mathbf{HO_{3Beco}}$ **and** $\mathbf{HO_{3Ceco}}$ that the continued implementation of RBT and FFF does not increase the availability of fish food organism in LCR. During MWF managed flows, the discharge from HLK were significantly stabilized with much lower discharges during the first half of the flow period. These lower flows result in less wetted habitat and reductions in velocity, both of which are expected to decrease the availability of fish food. In contrast, the flows during RBT and FFF are more stable, with the peak flows and lowest flows eliminated. These flows maintain a greater area of wetted habitat and maintain higher velocities, both of which are expected to increase the availability of fish food organisms.

ACKNOWLEDGEMENTS

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External data has been provided by the following organizations: Ministry of Forests, Lands, Natural Resource Operations and Rural Development, City of Castlegar, Columbia Power Corporation, Zellstoff Celgar Limited Partnership and Poisson Consulting Ltd.

Kyle Hawes of Ecoscape has been a consistent field lead, always prioritizing the safety of the field crew and the rigor of data collection. The Ecoscape field crew has included Heather Larratt, Rachel Plewes, Robert Wagner, Mike Schutten, Mary Ann Olson-Russello, Tina Deenik, Courtney Barr, Jason Schleppe, Adam Patterson and Angela Cormano. Data munging and analysis has been provided by Rachel Plewes, Jordie Ackers, Jason Schleppe, Chloee Robertson, and Noel Swain. Misty Schleppe has provided consistent administrative support and Mary Ann Olson-Russello has endeavoured to keep this project on-time and within budget.

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1.0 INTRODUCTION

This was a twelve-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 near the southern British Columbia border. Discharges from the HLK Dam during winter and spring have the potential to affect salmonid spawning and rearing habitats. To minimize impacts, BC Hydro and Power Authority (BC Hydro) altered operations of HLK Dam to include: 1) rainbow trout protection flows which stabilize or increase HLK discharges from April 1 through June 30, to reduce redd dewatering and subsequent egg loss of rainbow trout, and 2) mountain whitefish flows which limit maximum discharges during peak spawning in January and later stabilizes discharges to reduce egg dewatering until mountain whitefish emerge in late March (BC Hydro 2007).

The objective of this study was to examine the influence of the managed flow periods (Mountain Whitefish (MWF) Jan 1 - Mar 31; and Rainbow Trout (RBT) Apr 1 - Jun 30; and a control fall fluctuating flow (FFF) Sep 1 - Oct 31) period on select physical habitat components and ecological productivity measures. Benthic productivity, inclusive of periphyton and benthic invertebrates, are a primary food source for fish. Physical habitat components are important variables that influence the benthic productivity of a river.

The Physical Habitat component involved monitoring water temperature, stage, electrochemistry and nutrient levels in LCR to allow tracking of potential changes in physical habitat and ecological health due to flow conditions. The Ecological Productivity component involved monitoring periphyton and benthic invertebrates to assess potential changes in trophic productivity and overall ecological health of LCR resulting from the continued implementation of MWF, RBT and FFF (BC Hydro, 2005a,b).

This final report provides a summary of hydrological and benthic productivity data collected between 2008 and 2019. It addresses the CLBMON-44 management questions and hypotheses.

2.0 STUDY AREA

The study area is in southeast British Columbia on LCR between HLK Dam and the BBK gauging station. 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 (Figure 2-1).

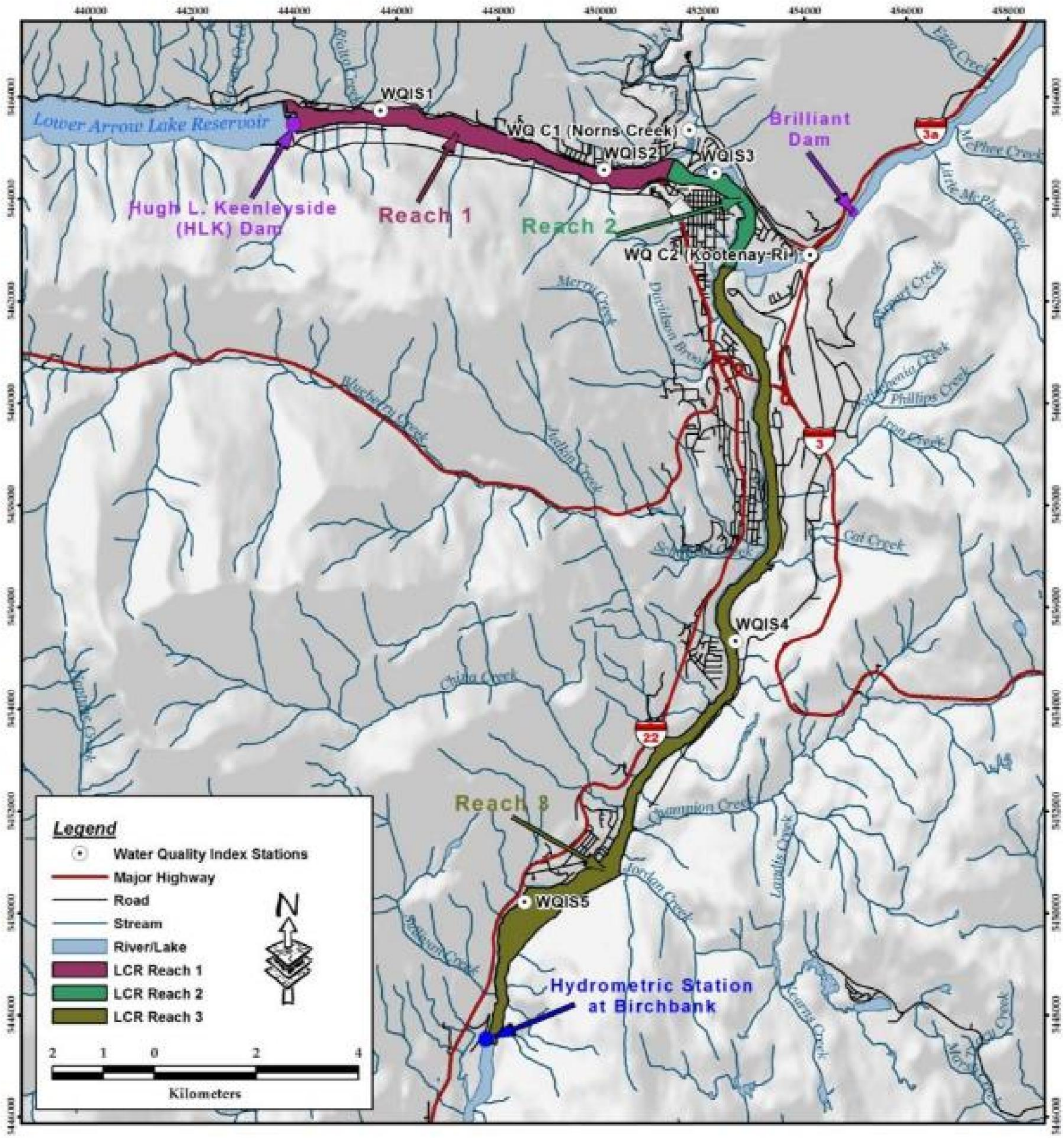

Figure 2-1: Map of the Lower Columbia River study area.

3.0 METHODS

3.1 Overview

There were two types of monitoring stations, water quality index stations (WQIS) and benthic productivity sampling stations (Table 3-1). Physical parameters collected at WQIS included water temperature, water level and water quality data. The locations of WQIS are depicted on Figure 2-1. There were five WQIS(1-5) on LCR and a single station on Kootenay River (WQC2).

Level loggers were installed at each of these sites to allow for continuous collection of water temperature and water stage (elevation). Water quality sampling was undertaken four times annually from 2008 – 2014 at the WQIS and at select tributary locations. Detailed methods pertaining to the collection of physical data (water temperature, stage and electrochemistry are provided in Appendices 3-5.

The benthic productivity sampling was mostly undertaken within Reach 2 of LCR between Norns Creek and Kootenay River (Figure A26). Periphyton and macroinvertebrate productivity monitoring was took place from 2008 - 2010, followed by alternating years (i.e. 2012, 2014,…2018). Productivity samplers were typically placed at seven sites within Reach 2 during summer, fall and winter sessions that ranged between 6 and 12 weeks in duration. The number of samplers at each site ranged from three to seven, with samplers placed at increasing depths from approximately 0.5 – 6 m. A typical design of the periphyton and macroinvertebrate sampling apparatus is shown in Figure 3-1, although different derivations of the apparatus were used at various points of the study.

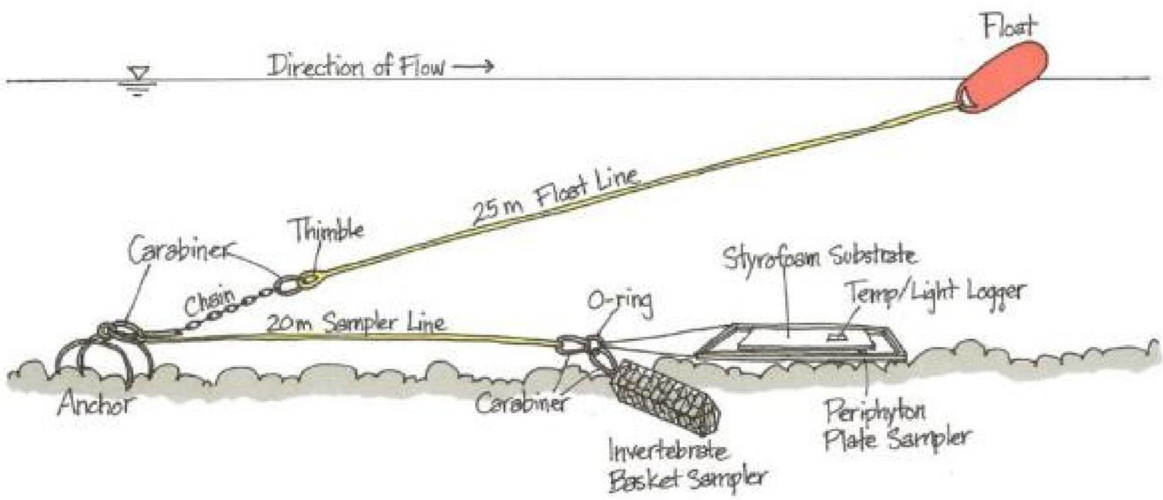

Figure 3-1: Typical design of the periphyton and macroinvertebrate sampling apparatus

During summer and fall 2008 - 2010, periphyton accrual sampling was undertaken to investigate chlorophyll-a biomass accrual rates. This sampling was also undertaken in winter 2014, 2016 and 2018. Deployed samplers were retrieved, sampled and then returned to the river at weekly or biweekly intervals during the 6 to 12-week deployments. Refer to Appendix 6 for accrual sampling results.

At the end of the deployment sessions, periphyton Styrofoam punches were randomly collected from each sampler to assess 1) chlorophyll-a; 2) Ash-Free Dry Weight (volatile solids) /total dry weight; and 3) taxa and biovolume. Benthic invertebrate baskets were also retrieved following standard protocols. Individual rocks from each basket were scrubbed to release clinging invertebrates. The contents from each basket were captured on a sieve and fixed with an ethanol solution, prior to transport to a laboratory for taxonomic identification and determination of biomass and associated metrics. More detailed information is available in Appendices 6-7.

Table 3-1: Monitoring Stations, Sample Types and UTM Coordinates (UTM 11).

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

Water quality sampling was also undertaken at additional tributary sites in early years of the study (Olson-Russello et al. 2012). Mainstem sites are mapped in Figure 2-1. Productivity sites are depicted in Appendix 6, Figure A27.

3.2 Datasets

The primary datasets collected as part of the CLBMON-44 study are separated as physical and productivity data and are summarized in Table 3-2. Additional data was used in the analyses for the various management questions; these data are included in appendices.

Table 3-2: Predominant physical and ecological productivity datasets.

Name/Description	Source	Years Obtained
Physical Datasets		
LCR / Kootenay River Elevation / Water Temperature	Data collected at 5 stations (LCR) and 1 station (Kootenay River)	LCR - 2008 – 2019* Kootenay – 2011 – 2019*
Mean Daily Discharge at Hugh L. Keenleyside (HLK), Brilliant Dam (BRD), and at Birchbank (BIR)	Data obtained from Poisson Consulting	2008 - 2018
Water Quality Parameters (electrochemistry and nutrients)	Data collected at 5 stations on LCR, and as many as 5 tributary creeks.	LCR - 2008 – 2015 Norns Creek / Kootenay River 2008 – 2015 Blueberry/China/Champion creeks 2008 - 2011
Productivity Datasets		
Light / Temp	Data collected at each productivity sampler during each deployment session	2008 – 2010, 2012, 2013(winter only), 2014, 2016, 2018
Benthic Invertebrates	Data collected at each productivity sampler during each deployment session. Data includes abundance, biomass, taxonomy and metrics	2008 – 2010, 2012, 2013(winter only), 2014, 2016, 2018
Periphyton	Data collected at each productivity sampler during each deployment session. Data includes abundance, biovolume, taxonomy and chlorophyll-a	2008 – 2010, 2012, 2013(winter only), 2014, 2016, 2018
Chlorophyll-a Time Series	Data collected at a select number of productivity samplers throughout the deployment periods	2008 – 2010 (summer and fall), 2014, 2016, 2018 (winter only)
Velocity	Data collected at each productivity sampler twice per deployment period	2009 – 2010, 2012, 2013(winter only), 2014, 2016, 2018
Substrates	Substrate percentage at each deployment site estimated during each deployment period	2009 – 2010, 2012, 2013 (winter only), 2014, 2016, 2018

*There are only partial datasets for some years.

3.3 Report Scope and Synopsis of Program Direction

This final report for CLBMON44 culminates 12 years of research. The report includes an Executive Summary with an overview status table of the management questions. The Introduction, Study Area and Methods sections provide a synopsis and context for the study. For each management question, a summary of important results are presented in Section 4, while additional supporting information is offered in Appendices 3 - 8; a separate appendix is provided to address each management question, and is structured as a brief stand-alone report.

CLBMON44 management questions covered biologically significant elements of the LCR that underpin its fisheries. None of the original management questions were eliminated from the CLBMON-44 program, however, it was acknowledged that not all management questions could be statistically addressed, and in some cases, resources were diverted because of it. For example, originally CLBMON-44 was designed with two productivity sampling periods, summer and fall. However, without a winter productivity sampling session, it was impossible to describe benthic productivity of LCR during the MWF flow period. Transitioning to sampling winter productivity and chl-a accrual rates proved to be crucial because periphyton productivity was surprisingly high during MWF flows. However, to incorporate this information while staying within the designated budget, the 4-5 annual water quality sample collections were suspended after 2015.

The core challenge of this study occurred because data collection began after MWF, RBT and FFF periods were established. Therefore, a typical pre / post study design could not be undertaken. For some management questions, such as *Physical Management Question #2, How did the various flow periods affect seasonal and inter-annual range and variability in river level fluctuation in LCR*, we were able to predict river elevations prior to the managed flow period using an established relationship between HLK discharge and river elevation. However, prediction was not possible for the Productivity Management Questions, and therefore lines of evidence, including modeling, descriptive conditions/metrics, current literature and professional judgement were used to address each management question. We also searched for managed fish flow periods that resembled unmanaged flows and compared those results to the corresponding typical fish flow period. We acknowledge that an alternative study design comparing before/after data would have better addressed the management questions, however we have attempted to address each question as thoroughly as possible, given the constraints of the existing data.

4.0 MANAGEMENT QUESTIONS

4.1 Flow Context

All CLBMON-44 management questions focus on how MWF, RBT and FFF originating from the HLK Dam affect various physical and productivity measures. Figure 4-1 illustrates the annual hydrograph of LCR below HLK Dam and Kootenay River below Brilliant Dam. The combined mean daily flow of LCR and Kootenay River are also shown at the Birchbank gaging station. The data is presented as all years of the study combined (2008-2018) and as 2018 alone (as this data had yet to be reported). Each of the managed flow periods are also depicted on Figure 4-1 and in Table A2, so the reader can visualize typical operations compared to 2018 during each period. Although the management questions focus on flows originating from HLK, the Kootenay River plays a significant role, as it backwaters upstream of its confluence into Reach 2 where productivity sampling largely took place. The flows at HLK and Brilliant dams are also frequently inversely managed (i.e. when HLK flows are high, Brilliant flows are low) (Figure 4-1 and Table A3).

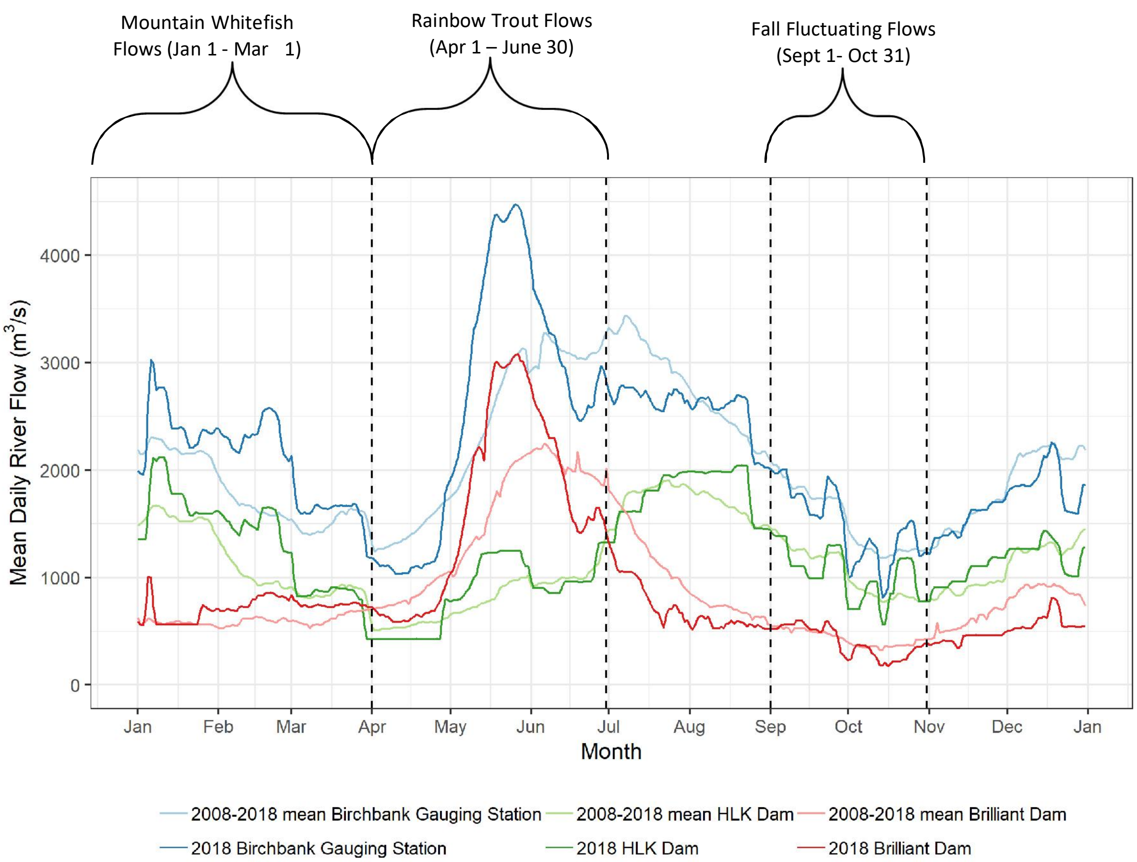

Figure 4-1: Mean daily river flow at HLK Dam (Columbia River), and Brilliant Dam (Kootenay River) in 2018 compared to the mean daily flow for all years (2008 – 2018).

Overall, the flows and water levels during the 2018 study period were higher than the mean daily flow for all combined years (2008-2018). The highest flow recorded at the Birchbank gauging station in 2018 was 4,473.9 m^3/s on May 26th, which was higher and earlier than typical. LCR flows normally peak in early July and the mean peak flow for all combined years was 3,483 m^3/s.

BC Hydro adopted an adaptive management plan to address fish stranding impacts and to protect spawning and rearing. During the MWF flow period (Jan 1 – Mar 31), averaged flows prior to management exceeded 1500 m^3/sec for the first half of the flow period, while managed flows remained below that threshold. Throughout the flow period, managed flows were less variable and lower than unmanaged flows. During the RBT flow period (Apr 1 – Jun 30), averaged flows prior to management were far more variable, while managed flows were more consistent. With management, flows less than 600 m^3/sec were eliminated by redistributing April flows and by reducing freshet flows and releasing that water gradually after peak freshet had passed. The RBT flow period results in a steadily increasing hydrograph that limits dewatering of the shallows. Throughout the FFF period (Sep 1 – Oct 31), managed flows were less variable than unmanaged flows. Averaged flows prior to management fell below 1100 m^3/sec for the first half of the flow period, while managed flows stayed above that threshold.

There was substantial variability in the managed flows between years during the MWF, RBT and FFF periods when productivity sampling was undertaken (Figure 4-2, Figure 4-3, Figure 4-4). For example, MWF flows during 2013 were held at a much higher discharge for the first half of the flow period. Atypical flows during 2016 also ended much lower than usual (Figure 4-2). Similar variability was also documented in 2016 for RBT flows (Figure 4-3) and in 2012 for FFF (Figure 4-4). This variation is useful in the explanation of some of the water quality, periphyton and benthic invertebrate productivity measures (Refer to Appendices 5-8 for additional discussion).

Flow variability measures were calculated for years and flow periods when periphyton and benthic invertebrates were sampled. The flow variability measure for the MWF is the elevation drop between spawning and incubation. The winters of 2013, 2016 and 2018 had similar elevation drops of 2.26-2.74 m, whereas 2014 had a much lower elevation drop (Table A4). For the RBT flow period, the sum of elevation drops (elevRBT) was used as the flow metric. The range of elevRBT was 0.640-2.6 m for all years when productivity sampling was conducted. The high flow year of 2012 had the highest elevRBT, whereas 2008 had the lowest elevRBT (Table A5). The mean daily standard deviation of flow (flowSD) was used as the flow metric for the FFF period. The falls of 2012, 2014, 2016 and 2018 had higher daily flow variability compared to 2008-2010 (Table A6).

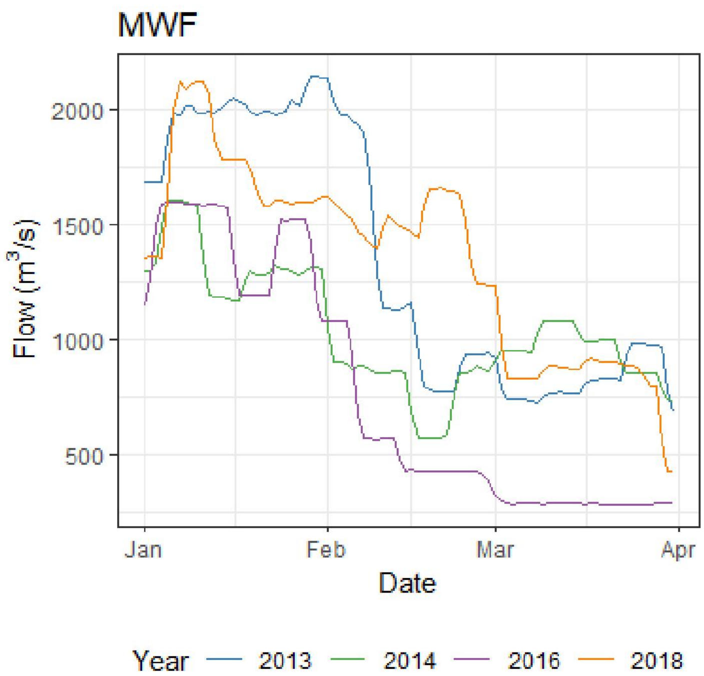

Figure 4-2: **Mean daily discharge from Hugh Keenleyside Dam during the Mountain Whitefish (MWF) flow period in years when winter productivity sampling was undertaken.**

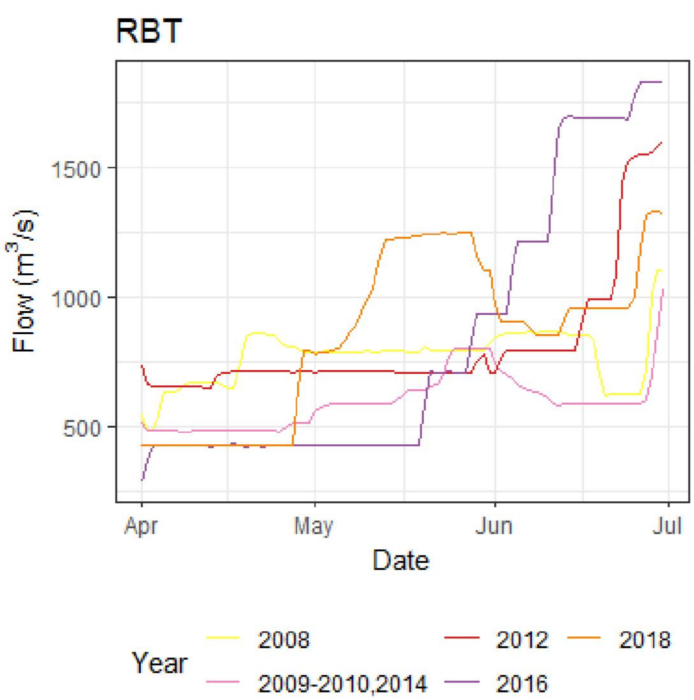

Figure 4-3: **Mean daily discharge from Hugh Keenleyside Dam during the Rainbow Trout (RBT) flow period in years when summer productivity sampling was undertaken.**

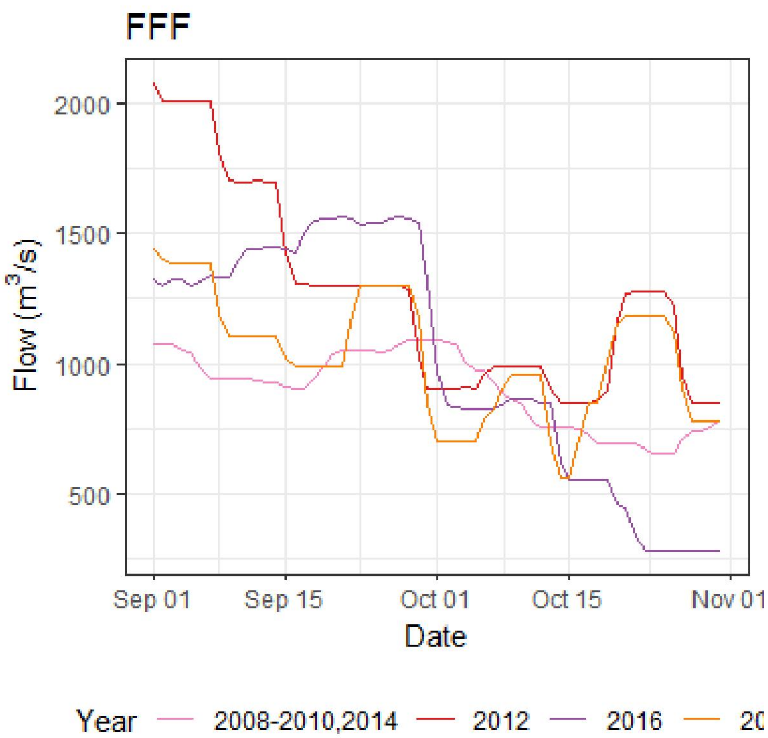

Figure 4-4: Mean daily discharge from Hugh Keenleyside Dam during the fall fluctuating (FFF) flow period in years when fall productivity sampling was undertaken.

Figure 4-5, Figure 4-6, and Figure 4-7 depict the pre-managed regime for each flow period compared to the managed flows. While the FFF period is theoretically a control for the MWF and RBT flows, the flows are considerably more stable during managed years. The similarity of MWF flows in 2013 (Figure 4-2 and Figure 4-5) with the pre-MWF flows (Figure 4-5) offer a potential for before/after comparisons. This similarity allows us to infer that productivity data collected during the winter of 2013 may have similarities to river productivity prior to the implementation of managed fish flows.

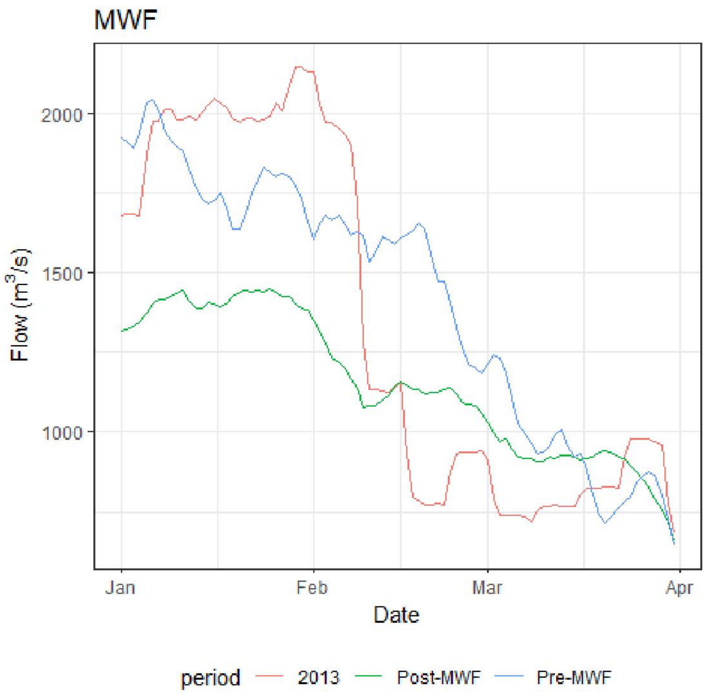

Figure 4-5: Mean daily discharge from Hugh Keenleyside Dam for years sampled Pre and Post Flow Management during Mountain Whitefish (MWF) Period.

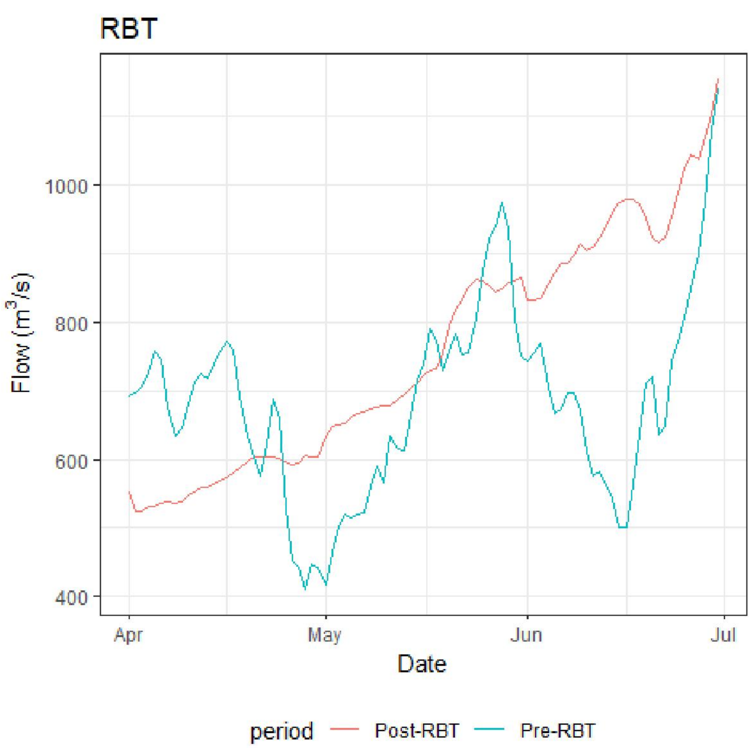

Figure 4-6: Mean daily discharge from Hugh Keenleyside Dam for years sampled Pre and Post Flow Management Pre and Post Flow Management during Rainbow Trout (RBT) Period.

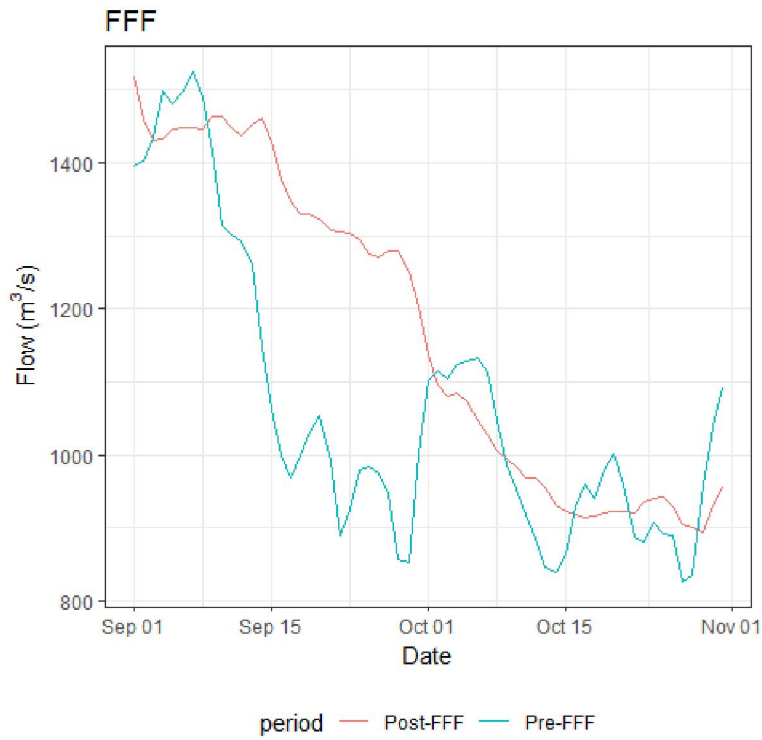

Figure 4-7: **Mean daily discharge from Hugh Keenleyside Dam for years sampled Pre and Post Flow Management during Fall Fluctuating Flow (FFF) Period.**

4.2 Physical Habitat Management Questions

4.2.1 Physical MQ1

Physical MQ #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?*

Physical management question #1 is best addressed with Figure 4-8. During the study, the mean daily water temperatures at the five LCR WQIS varied seasonally ranging from approximately 3 to 18°C. Temperatures in Kootenay River (WQ C2) were slightly higher, and ranged from about 3 to 20°C. Water temperatures follow a seasonal pattern. During the MWF flow period (Jan 1 – Mar 31), water temperatures were consistently between 2.5 and 5°C. Temperatures during the RBT flow period (Apr 1 – Jun 30) steadily increased from approximately 5 to 15°C. Finally, the FFF period exhibits the opposite trend with water temperatures declining from approximately 18 to 10°C (Figure 4-8).

The 2018 daily water temperatures were generally similar to the mean water temperatures recorded throughout the study (Figure 4-8). WQIS4 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) and Plewes *et al.* (2017) 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, resulting in river temperatures with predictable spatial patterns.

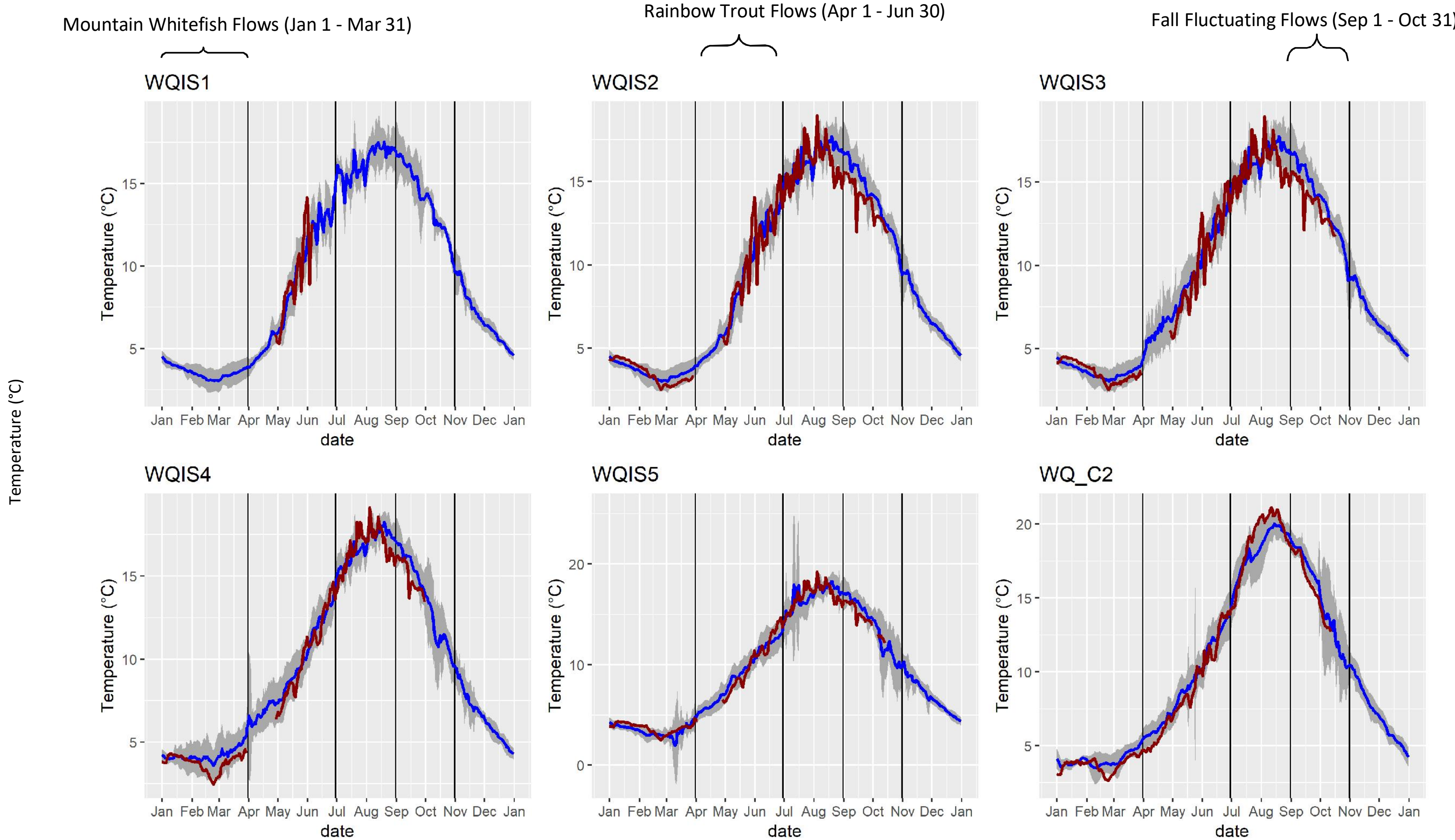

Figure 4-8: **Mean daily water temperatures recorded at WQIS1 – 5 on LCR and at WQ C2 on Kootenay River.** The red line depicts the mean daily water temperature recorded at each site in 2018. The blue line is the mean daily water temperature throughout the duration of the study (2008-18) ± SD (gray shaded area). The vertical lines indicate the beginning and end of each flow period.

To test the specific hypotheses that implementation of different flow periods may affect LCR water temperature, we ranked the relative importance of flow regime with other parameters that can affect water temperature including reservoir temperature, reservoir elevation, year and air temperature. The analysis was undertaken for two sites, WQIS1, which is upstream of Kootenay River, and WQIS5, which is downstream of Kootenay River. At WQIS1, LCR water temperature was most strongly correlated with Arrow Lakes reservoir temperature during the MWF flow period, and Arrow Lakes reservoir elevation during the RBT and FFF periods (Figure 4-9).

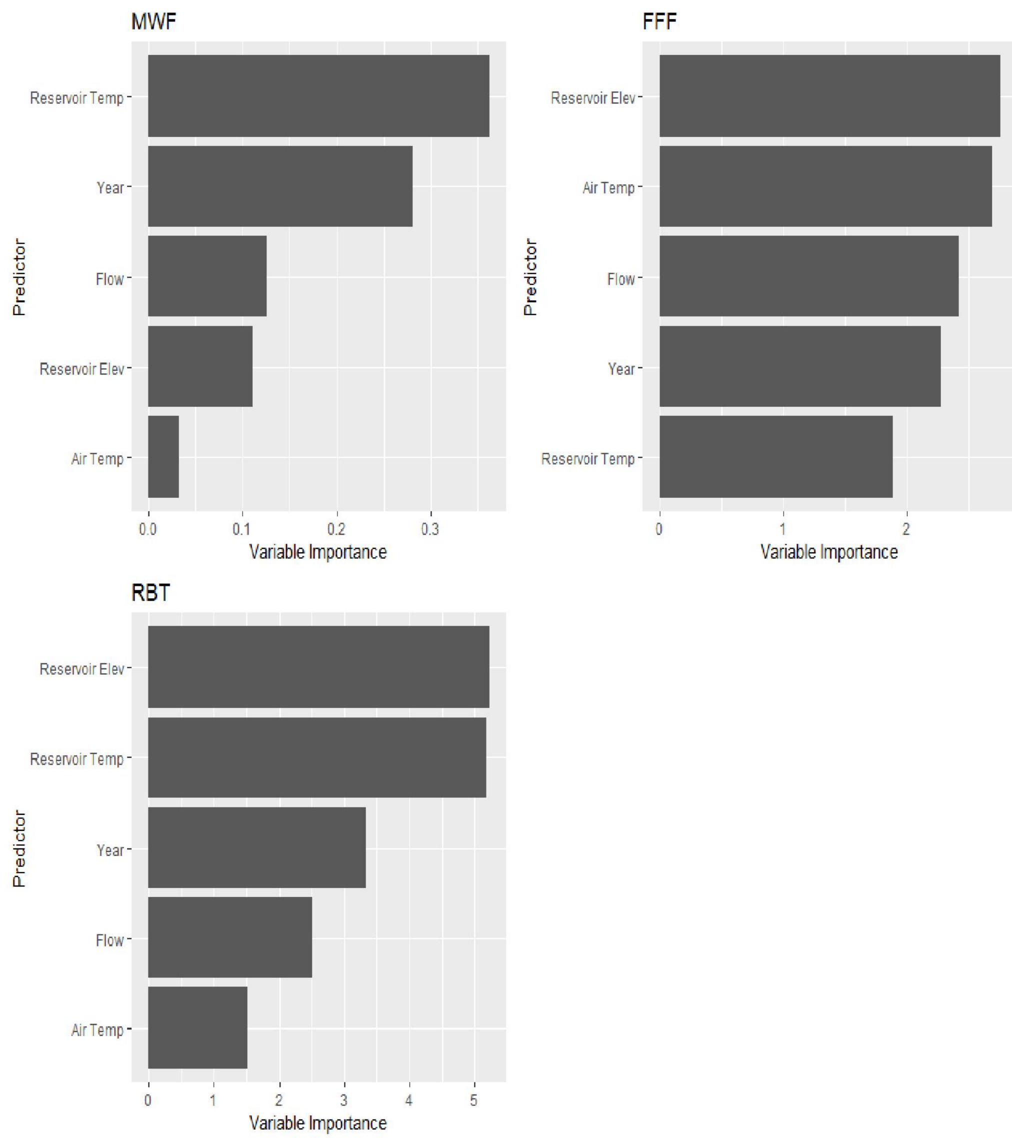

Figure 4-9: Random Forest Model variable importance plots for LCR water temperature during MWF, RBT and the FFF periods at WQIS1.

Although flow played a role, it was never a critical predictor of LCR water temperature at WQIS1 (Figure A2, Figure A3 and Figure A4). Additional results pertaining to diel, spatial differences in water temperature, and the influence of MWF, RBT and FFF on water temperature are presented in Appendix 3.

4.2.2 Physical MQ2

Physical MQ #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?*

This question is best addressed with Figure 4-10. It shows the mean daily water levels during each flow period at five stations on LCR (WQIS1-5) for all years of the study and for one station on Kootenay River for eight years of the study. Data from 2018 is depicted separately, as it had not yet been reported. Water levels during 2018 differed from most years of the study and are further discussed in Appendix 4.

Over the 12-year study, the water levels have differed slightly from year to year, but the managed flows have generally resulted in a consistent pattern. During MWF flows (Jan 1 – Mar 31), LCR water levels decline approximately 1.5 m. During RBT flows (Apr 1 – Jun 30), LCR water levels steadily increase and typically peak just after the flow period ends in early July. The change in water elevations during this period can approach 2.5 to 3 m. During the control FFF (Sept 1 – Oct 31), LCR water levels gradually decline approximately 1 m over the two-month period.

Although there are variations in the magnitude of water levels from year to year, the managed flows are intended to:

- limit maximum discharges during peak mountain whitefish spawning in January and stabilize discharges during the remainder of the flow period to reduce egg dewatering until mountain whitefish fry emerge in late March; and
- During RBT flows, the discharges increase from April 1 through June 30, to reduce redd dewatering and subsequent egg loss of rainbow trout.

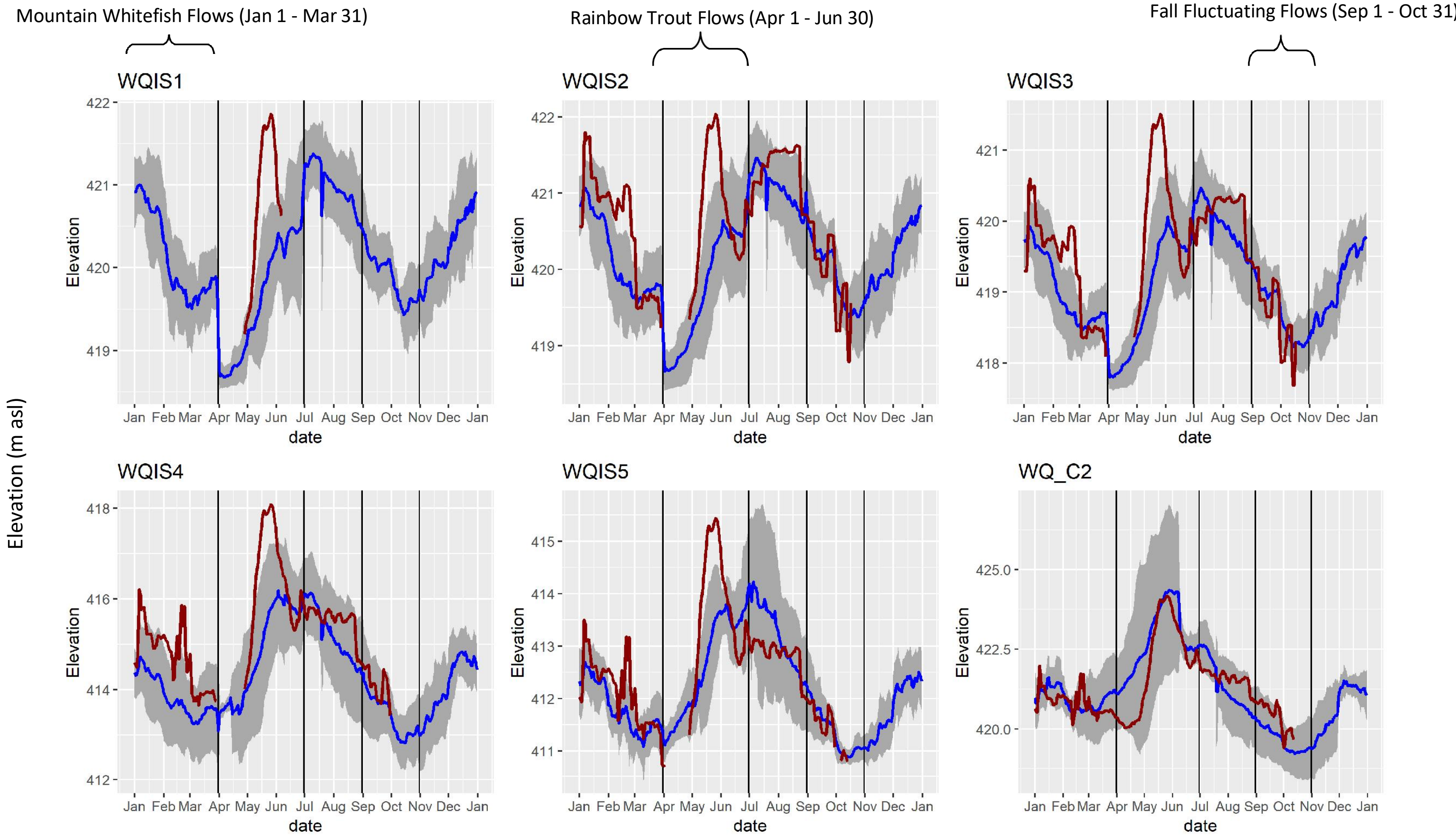

Figure 4-10: Mean daily water levels recorded at WQIS1 – 5 on LCR and at WQ C2 on Kootenay River. The red line depicts the mean daily water level recorded at each site in 2018. The blue line is the mean daily water level throughout the duration of the study for LCR sites (2008-2018± SD (gray shaded area)) and for an eight-year duration at the Kootenay River site (2011 - 2018± SD (gray shaded area)). The SD is shown to highlight the variation in the data over multiple years, but it could not be determined for all months due to gaps in the data.

4.2.3 Physical MQ3

Physical MQ #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?*

Based on data collected throughout the study, LCR has good water quality and limited biologically available nutrient concentrations indicative of oligotrophy. Parameters rarely exceeded water quality guidelines or objectives.

The biologically available nutrient data indicated that nitrate + nitrite and SRP concentrations were capable of influencing periphyton production in the LCR. Fish flows (MWF, RBT and FFF) may improve particulate and dissolved nutrient delivery under stabilized, less variable flow conditions relative to unmanaged flows, but they are unlikely to alter the overall nutrient status of LCR. For example, winter low flows frequently had greater nutrient concentrations and greater periphyton productivity than high flow periods.

Fish flows caused small decreases in electrochemistry parameters through dilution, and pH was stable throughout the flow periods. Dissolved oxygen increased during high flow periods but would not affect the periphyton community.

We conclude that the continued implementation of MWF, RBT and FFF periods has an affect on the electrochemistry and biologically active nutrients in LCR. Lines of evidence to support this conclusion include:

- The comparison of 2013 MWF flows (similar hydrograph to unmanaged flows) with 2014 and 2015 showed elevated conductivity and nitrate concentrations in 2013.
- Descriptive statistics suggested that MWF managed flow periods may influence total phosphorus concentrations.
- Operations such as Celgar and sewage outflows can increase the range of T-P values observed during winter low flows (low dilution), and this concentration effect would be evident in the first half of the MWF flow period.
- T-N results indicate that there are additional nitrogen sources in LCR that augment concentrations in the flows from ALR above its confluence with Kootenay River. These sources would experience less dilution with managed MWF flows than unmanaged flows over the same period.
- Descriptive statistics indicated that flow variability increased the availability of some nutrients (NO_2+NO_3 and total P), and RBT flows restrict flow variability.
- Descriptive statistics suggested that RBT managed flow periods may influence total phosphorus concentrations.

- The comparison of 2012 RBT (extreme freshet) to the other RBT flow years showed significantly lower conductivity, higher nitrate and lower T-P concentrations compared to other RBT flow periods.
- The stabilized RBT flows should lower organic and total nitrogen concentrations by reducing scour.
- Turbidity and TSS are positively correlated with flows, so lower peak velocities in RBT flow periods would limit turbid conditions.
- Substrate exposure occurs during very low flows and can affect rates of groundwater influx and organic decay, particularly at shallow and mid shallow sites and substrate exposure was reduced under the first half of the fall fluctuating flow period.
- The higher averaged flows in the first half of the FFF period should decrease electrochemistry concentrations through dilution but increase organic and total nutrient concentrations through scour.

4.2.4 Challenges in Addressing Physical MQ3

Water quality data collection of 4-5 events per year is common practice but for this study, this frequency prevented many statistical analyses. Because of this, *Physical Management Question #3; How did the various flow periods affect electrochemistry and biologically active nutrients,* had to be addressed using multiple lines of evidence and professional judgement only.

The initial years of this study (2008 -2010) involved productivity sampling in Reaches 1,2 and 3. The study design was then revised to focus subsequent productivity sampling in Reach 2 only, but the water quality sampling stations were retained to provide an overview of the LCR system, and only two of the stations occurred in Reach 2. This restricted opportunities for correlating LCR water quality with its benthic productivity.

The absence of data collection prior to managed fish flows prevented a typical pre / post study design. Instead, we employed the lines of evidence approach. This included comparing water quality for managed fish flow periods that resembled unmanaged flows to those from the corresponding typical fish flow periods.

4.3 Ecological Productivity Management Questions

4.3.1 Challenges in Addressing Productivity Management Questions

Addressing how managed flows affect the benthic invertebrate and periphyton communities is challenging because the management questions cannot be statistically tested directly. Because this monitoring program began after the MWF and RBT managed flows were initiated, there is no pre-managed flow productivity data. And, it is not possible to predict or model productivity without some pre-flow change data.

Previous annual reports explored the use of managed flow derivatives (e.g. Elev Diff (MWF), Flow Daily SD) as predictors in mixed effects models of productivity. However, these flow measures were highly correlated with general measures of flow and the use of the flow derivatives violated statistical assumptions due to pseudo replication within a given season and year. The limited variation in MWF, RBT and FF flow derivatives prevented our ability to detect an actual effect.

We therefore had to address the ecological productivity management questions by looking at multiple lines of evidence, including:

- how the managed flows varied over the years of study (e.g. Figure 4-2 to Figure 4-4),
- identifying potential relationships between managed flow and benthic periphyton and invertebrate community structure,
- identifying seasonal shifts in the benthic invertebrate community and determining if those shifts could be related to managed flows, and
- relying on professional judgement and available scientific literature of periphyton and benthic invertebrate community composition below dam impoundments.

4.3.2 Productivity MQ1

Productivity MQ #1: *What is the composition, abundance, and biomass of epilithic algae in LCR? 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 epilithic algae?*

Periphyton sampling was focused on the most productive area of the river - the permanently wetted, shallow substrates in LCR Reach 2, all in the photic zone from the water's edge to depths of 5 - 6 m. Sites that experienced frequent dewatering of their shallow and mid-shallow samplers (S2, S3, S4) had lower productivity and diversity, particularly in the FFF period. The transect depth where peak biomass occurred varied with season and sample site.

LCR periphyton community composition was stable within each season, and diatom dominated. Diversity metrics were similar over the transect span from shallow to deep. However, there were substantial differences in the periphyton community observed between the three seasonal deployments in LCR. Periphyton diatom guilds were determined from the literature and expanded to include soft-bodies algae using algae morphology (refer to Appendix 11.4). Taxa from the low-profile guild (taxa that can withstand higher flows) contributed more to periphyton community in the high flow seasons of summer and fall when water temperatures were warmer, while the percent high profile guild (taxa vulnerable to high flows) was more abundant in winter during cooler water temperatures and stable flows. In the winter, sites closest to HLK (S1 and S2) had a low percent biovolume of high-profile guild because a large proportion of the biovolume was from planktonic diatoms. Over the years of study, the largest shifts in community structure occurred in the soft-bodied algae such as flagellates, filamentous green algae and cyanobacteria (Figure 4-11, Table A19 Table A20).

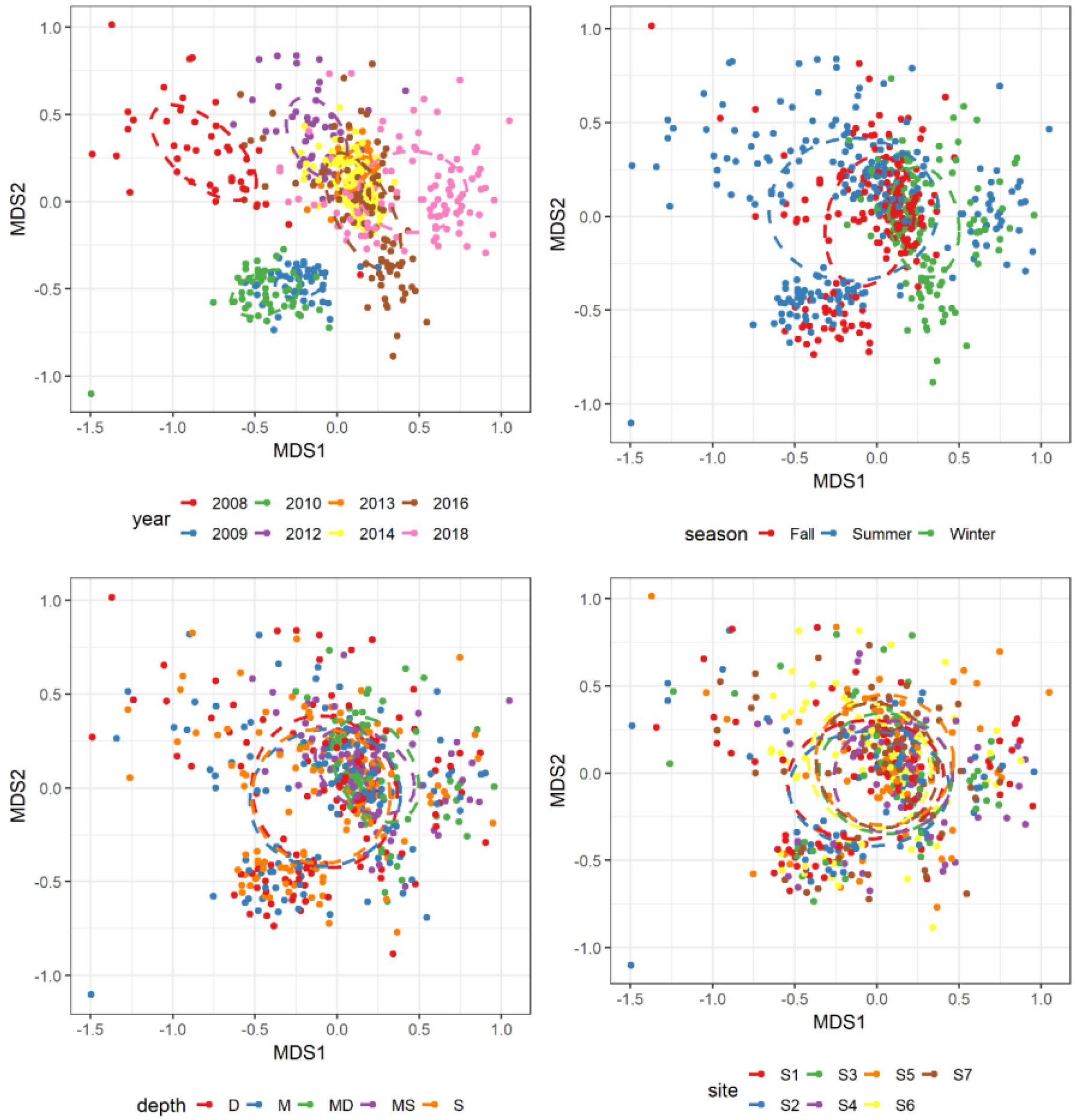

Figure 4-11: NMDS of periphyton family level abundance grouped by season, depth, year and site for all data from 2008 – 2018. Closer points indicate a more similar periphyton community composition. The NMDS used a Bray-Curtis dissimilarity index and had a stress index of 0.23. Ellipses are calculated based on 95% confidence interval of the NMDS scores for each group.

Periphyton standing crop metrics of total biovolume, chl-a, and total abundance were lowest in the summer period which includes freshet and highest in the winter – in part because of the winter Didymo proliferation, particularly at sites with back-watering from the Kootenay River (Figure 4-12). Of the seven sample sites in R2, S6 consistently had the lowest periphyton productivity across all seasons, particularly in winter and fall when other sites were far more productive, and it was the only permanently depositional site in all flow regimes (Figure 4-12).

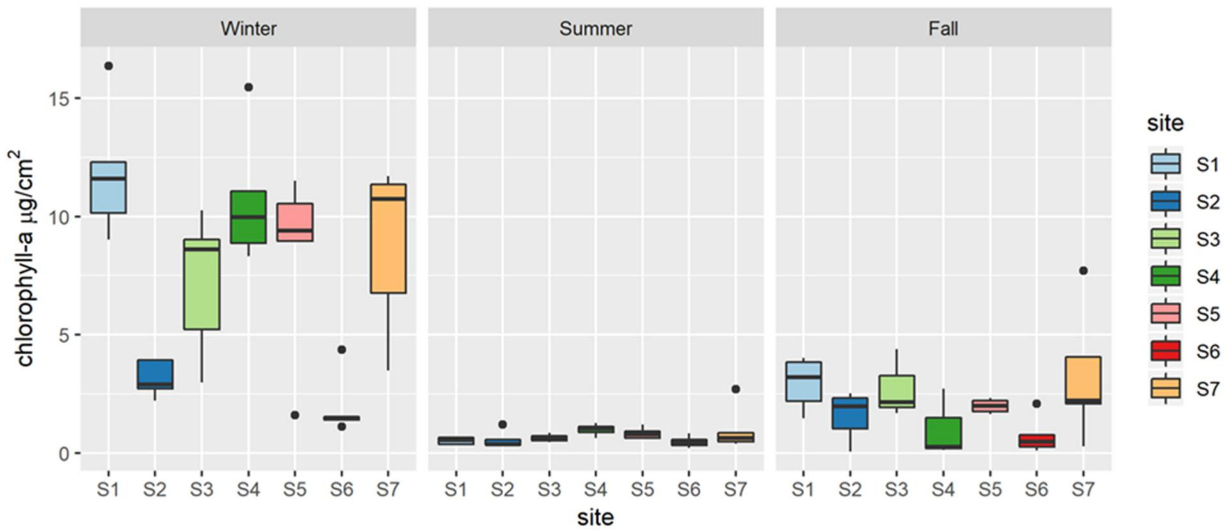

Figure 4-12: **Chlorophyll-a (ug/cm2) biovolume (cm3/m2) and chl-a (ug/cm2) in winter, summer and fall in typical years 2014, 2016 and 2018, over the range of sampled depths.** Depth labels are: S=shallow, MS=moderately shallow, M=mid, MD=moderately deep, D=deep.

In the MWF flow winter period, lower water temperatures of 4 – 6°C and reduced light intensity coupled with shorter day length apparently exerted less influence than the benefits of stable flows because winter samplers showed higher overall periphyton production than in other flow periods; however, the time to achieve peak biomass was longer.

The lowest overall periphyton production and diversity were observed during the RBT summer flows when freshet was occurring. Shear and scour of periphyton from higher velocities during high flow periods are likely the cause of this observation. Specifically, high-profile filamentous green taxa and Didymo masses can be dislodged with small increases in velocity above 0.2 m/sec, while tightly attached low-profile diatoms require increased shear stresses to experience the same scour.

The moderate flows during the FFF period allowed more periphyton growth compared to the summer, resulting in a relationship between season and production. Across all years, periphyton productivity increased during the fall at most sampled depths, except in several shallow sites. Periodic dewatering of shallow substrates along the water's edge reduced their fall periphyton production and increased mortality.

Finally, the LCR chl-a time series data indicated that periphyton accrual reaches peak biomass in 6-7 weeks in summer, needs longer than 8 weeks in fall and longer than 10 weeks in winter (Figure 4-13). LCR accrual rates were apparently slower in winter than in summer and fall, suggestive of temperature effects.

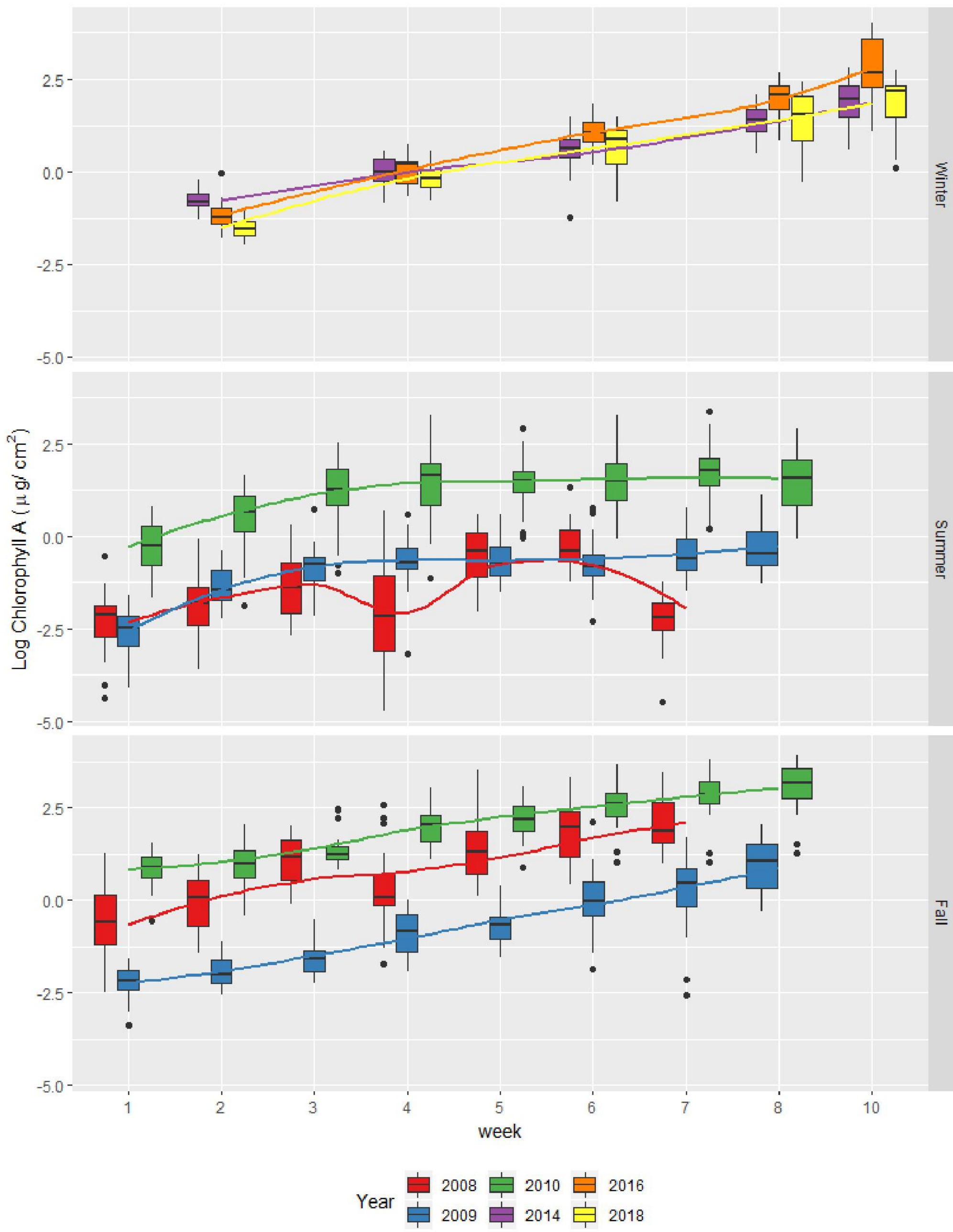

Figure 4-13: Weekly time series periphyton chl-a accrual rates in summer (2008 – 2010), fall (2008 – 2010) and winter (2014, 2016, 2018). Fitted lines were generated using a locally weighted polynomial regression method (LOWESS). The first three years of data were obtained from Scofield et al. 2011.

In summary, hydraulic conditions in general and managed flows in particular can influence periphyton community structure, its biomass and its accrual rate. Additional periphyton analyses and results are provide in Appendix 6.

4.3.3 Productivity MQ2

Productivity MQ #2: *What is the composition, abundance, and biomass of benthic invertebrates in LCR? 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?*

The abundance and biomass of sampled benthic invertebrates in LCR across all years and seasons are shown in Figure 4-14 and Figure 4-15, respectively. The total number of benthic invertebrate samples (n) collected during the study in summer, fall and winter was 217, 170 and 133. The highest mean abundance (#/basket) ± SD occurred in the summer with 6,674 ± 6,890 organisms per basket, followed by fall and winter with 5,079 ± 5,077 and 3,441 ± 4,399 (Figure 4-14). Although summer maintained the highest mean abundance, fall samples consistently had the highest biomass (mg/basket±SD) (2.0 ± 3.2), followed by summer (1.2 ± 1.6 mg), and winter was typically lower (0.9 ± 1.5 mg) (Figure 4-15).

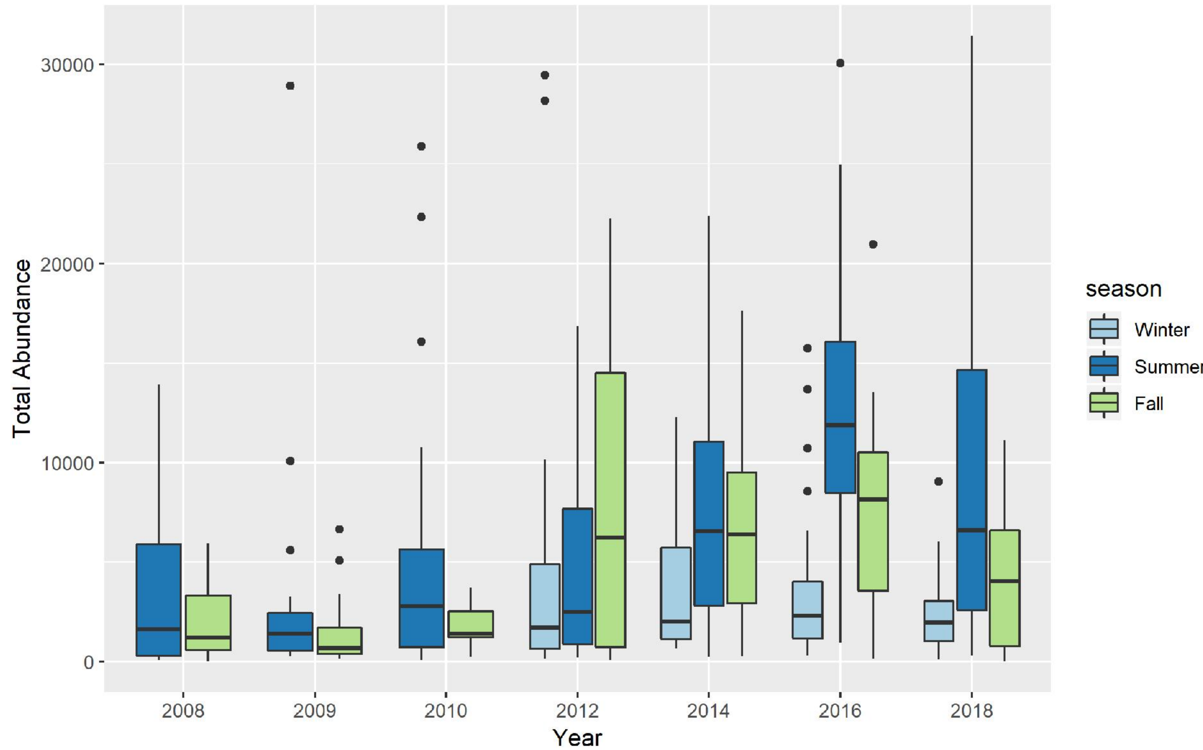

Figure 4-14: Total abundance of benthic invertebrates grouped by season and year.

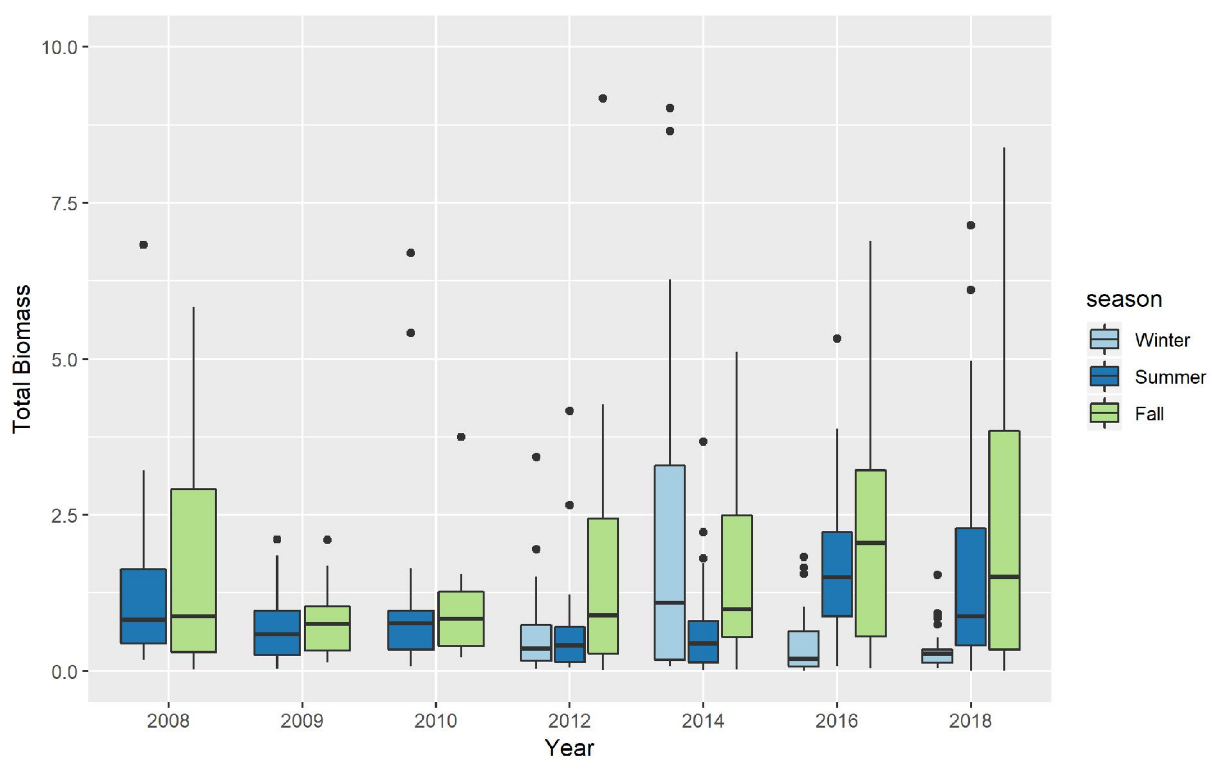

Figure 4-15: Total biomass (mg) of benthic invertebrates grouped by season and year.

The composition of benthic invertebrates varied by site, depth, season and year (Figure A42), with the most obvious shifts occurring between seasons (Table 4-1). Three families of benthic invertebrates comprised the majority of LCR samples. Chironomidae (nonbiting midges), Hydropsychidae (net-spinning caddisflies) and Simuliidae (black flies) made up together more than 86, 80 and 79 percent of invertebrates in winter, summer and fall. Hydropsychidae had high abundances in summer and fall and were barely documented during the winter, while the abundance of Simuliidae vastly increased in the winter, compared to summer and fall.

Table 4-1: Composition of benthic invertebrates at the family level across seasons.

	Winter	Summer	Fall
Benthic Invertebrate Family	Percent Mean of Relative Abundance ± (SD)	Percent Mean of Relative Abundance ± (SD)	Percent Mean of Relative Abundance ± (SD)
Chironomidae	53.2±26.9	28.9±14.8	43.0±22
Hydropsychidae	1.5±2.3	45.4±25.7	29.3±23.2
Simuliidae	32.1±31.9	7.3±11.1	7.8±13.6
Hydridae	5.5±8.8	2.8±6.5	8.9±17.2
Ephemerellidae	5.4±4.9	-	7.5±5.6
Trichoptera	-	6.3±17.3	-
Lymnaeidae	-	5.0±7.9	3.3±6.6
Lumbriculidae	3.4±5.2	-	5.0±8

-Families with minimal presence in a season.
Shaded families comprised at least 30% of the mean abundance in 1 or more seasons.

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 is further explored in Appendix 7.

4.3.4 Productivity MQ3

Productivity MQ #3: *Are organisms that are used as food by juvenile and adult MWF and RBT in LCR supported by benthic production in LCR?*

The stomach contents of 120 RBT and MWF caught in the fall of 2012 and 2014 were analyzed for benthic invertebrates. The dominant taxa for most fish stomachs were Hydropsychidae (net-spinning caddisflies, Trichoptera). On average the percent relative abundance of Trichoptera in juvenile and adult MWF were 98±4.0% and 86±31%, whereas in juvenile and adult RBT the mean percent relative of abundance of Trichoptera were 64±37% and 64±35%.

Although on average Trichoptera had the highest abundance in most fish stomachs, a few fish had a higher abundance of Simuliidae (black fly; Dipteran). There were 2 adult RBT, 8 adult and 1 juvenile MWF that had greater than 50% relative of abundance of Simuliidae. In general, differences in fish stomach contents based on year, fish species and age were minimal.

A few of the 80 fish caught in 2014 had distinct invertebrate gut contents. For example, the stomach contents of two adult RBT and one adult MWF were dominated by Corixidae (water boatmen, Heteroptera). Two juvenile and one adult MWF, also caught in 2014, had stomach contents with the highest abundance of Glossosomatidae (little black caddisflies, Trichoptera).

The stomach content analysis confirmed that MWF and RBT consume Trichoptera and Diptera, and these dominant taxa in fish stomachs coincided with the most abundant benthic invertebrates sampled during the fall. For example, Trichoptera made up 73% and 56% of the total biomass of benthic invertebrates in fall of 2012 and 2014, respectively. The fish appear to key into abundant, readily available taxa.

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6.0 APPENDIX 1. Timeline and Milestones of CLBMON-44

Table A1. Timeline and Milestones of CLBMON-44.

Year	Milestones
2008	Start of contract. Productivity sampling occurred in all 3 reaches during summer (36 sites; 6 weeks; July 14 – August 25), and only in Reach 2 during fall (21 sites; 8 weeks; August 26 – October 20). Samplers per site ranged from 3 to 7. Chlorophyll-a weekly sampling in summer and fall (Reach 2 only). Water quality sampling at 6 tributary sites and mainstem of LCR 4x between July and October.
2009	Productivity sampling occurred in all 3 reaches during summer (35 artificial substrates for 8 weeks), and only in Reach 2 during fall (21 artificial substrates for 8 weeks). Chlorophyll-a weekly sampling in summer and fall (Reach 2 only). Water quality sampling at 6 tributary sites and mainstem of LCR 4x between July and October. Underwater natural substrates photographed.
2010	Last year of TG Eco-Logic contract. Level loggers dismantled in November 2010 and returned to BC Hydro. Productivity sampling occurred in all 3 reaches during summer (35 artificial substrates for 8 weeks), and only in Reach 2 during fall (21 artificial substrates for 8 weeks). Chlorophyll-a weekly sampling in summer and fall (Reach 2 only). Water quality sampling at 6 tributary sites and mainstem of LCR 6x between June and November. Underwater natural substrates visualized with an underwater camera and substrate size determined.
2011	Contract awarded to Ecoscape. Off-year for productivity monitoring. Water quality sampling at 6 tributary sites and mainstem of LCR monthly between July and October. Level loggers re-established at the same collection locations in July. Additional level logger installed on Kootenay River.
2012	Extreme high freshet flows in 2012 resulted in the malfunctioning of level loggers; loggers and data lost. New loggers were installed in August. The high flow event also resulted in an extended summer deployment session (35 artificial substrates for 11 weeks). Summer deployment was delayed
2013	First season of winter productivity sampling (35 artificial substrates for 12 weeks) (data presented with 2012 data); no time series sampling. Otherwise an off year for productivity sampling. Water quality sampling undertaken 4x at tributaries (Norns Creek and Kootenay River) and LCR between April and November.
2014	Productivity sampling during winter (35 artificial substrates for 10 weeks), summer (35 artificial substrates for 10 weeks), fall (35 artificial substrates for 10 weeks). Bi-weekly time series sampling (chlorophyll-a) for winter session only. Water quality sampling undertaken 4x at tributaries (Norns Creek and Kootenay River) and LCR between March and October.
2015	Off year for productivity sampling. Water quality sampling program terminated to free up budget for additional winter productivity sampling. Collection of water temperature and river elevation data only.
2016	Productivity sampling during winter (35 artificial substrates for 10 weeks), summer (35 artificial substrates for 10 weeks), fall (35 artificial substrates for 10 weeks). Bi-weekly time series sampling (chlorophyll-a) for winter session only.
2017	Quarterly downloads of level logger data only (water temperature and elevation).
2018	Productivity sampling during winter (35 artificial substrates for 10 weeks), summer (35 artificial substrates for 10 weeks), fall (35 artificial substrates for 10 weeks). Bi-weekly time series sampling (chlorophyll-a) for winter session only.
2019	Contract extension to continue collection of water temperature / water elevation data, and river bathymetry.

7.0 APPENDIX 2. River Flows Supplemental Results

Flow within the study area is dominated by discharges from HLK Dam on the Columbia River and from Brilliant Dam on the Kootenay River. In 2018, contributions made to the mean daily river flows from the Columbia and Kootenay Rivers were typical, at 56.6% and 40.3%, respectively, of the total flows measured at the Birchbank gauging station. The remaining 3.1% was contributed by smaller tributaries such as Norns Creek and outfalls.

Table A2. Mean Daily Flows in 2018 by Designated Flow Period (m3/s).

Mountain Whitefish Flows (Jan 1 - Mar 31)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	90	90	90
	Minimum	424.7	555.1	1185.5
	Maximum	2117.7	1006.4	3030.6
2018	Median	1474.0	730.2	2236.8
	Arithmetic Mean	1348.8	716.6	2121.1
	Standard Deviation	419.5	99.8	412.7
	Coefficient of Variation	0.3	0.1	0.2
Rainbow Trout Flows (Apr 1 to Jun 30)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	90	90	90
	Minimum	424.4	585.7	1029.1
	Maximum	1328.9	3083.4	4473.9
2018	Median	903.1	1545.0	2612.9
	Arithmetic Mean	848.3	1662.8	2601.0
	Standard Deviation	314.4	869.7	1198.9
	Coefficient of Variation	0.4	0.5	0.5
Fall Fluctuating Flows (Sep 1 to Oct 31)				
Year	**Statistic**	**HLK/ALGS**	**Brilliant**	**Birchbank**
	N (days)	60	60	60
	Minimum	563.8	176.5	809.7
	Maximum	1438.3	603.2	2012.1
2018	Median	1013.2	373.9	1516.3
	Arithmetic Mean	1031.0	399.1	1494.0
	Standard Deviation	234.3	140.8	332.8
	Coefficient of Variation	0.2	0.4	0.2

Table A3. Mean daily river flows (m3/s) at HLK Dam, Brilliant Dam and the Birchbank gauging station in 2018 and across all years of the study.

Location	N (days)	Statistic	2018	2008 - 2018
HLK	365	Mean	1213.3	1130.3
		Min	424.4	144.8
		Max	2117.7	3258.0
		SD	440.1	494.3
Brilliant	365	Mean	864.5	887.3
		Min	176.5	0.0
		Max	3083.4	4224.1
		SD	654.4	616.2
Birchbank	365	Mean	2144.8	2061.0
		Min	809.7	43.3
		Max	4473.9	6043.1
		SD	787.2	801.9

Table A4. Elevation difference (m) between spawning (Jan 1 – Jan 21) and incubation (Jan 22 – Mar 31) during MWF for each year sampled.

season	year	metric	value
Winter	2013	elevMWF	2.740
Winter	2014	elevMWF	1.670
Winter	2016	elevMWF	2.340
Winter	2018	elevMWF	2.260

Table A5. The sum of elevation drops during RBT for each year sampled.

season	year	metric	value
Summer	2008	elevRBT	0.640
Summer	2009	elevRBT	1.210
Summer	2010	elevRBT	0.950
Summer	2012	elevRBT	2.600
Summer	2014	elevRBT	2.110
Summer	2016	elevRBT	1.450
Summer	2018	elevRBT	1.930

Table A6. The mean of standard deviation of daily flow from HLK during Fall Fish Flows (FFF) for each year sampled.

season	year	metric	value
Fall	2008	flowSD	4.000
Fall	2009	flowSD	9.480
Fall	2010	flowSD	7.170
Fall	2012	flowSD	16.640
Fall	2014	flowSD	21.450
Fall	2016	flowSD	17.630
Fall	2018	flowSD	20.270

8.0 APPENDIX 3. Physical Habitat - Management Question #1

8.1 Introduction

This appendix further addresses the physical habitat management question #1 and relevant hypotheses.

MQ#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.*

8.2 Methods

Water temperature data was collected in two ways throughout the duration of the project. First, level loggers were installed at WQIS (5 on LCR and 1 on Kootenay River) that recorded temperature data every ½ hour. Second, TIDBIT loggers were deployed on productivity samplers that collected water temperature data during productivity deployments in the winter, summer and fall. The level logger data was used to address how the various flow periods affected water temperature in LCR, and if there were spatial differences in the pattern of observed water temperatures. The TIDBIT data was used to address diel and seasonal water temperature differences.

Level loggers were originally installed by TG Eco-Logic LLC in 2008 but were dismantled at the end of 2010 due to their contract end. In July 2011, AquiStar® PT2X Smart Sensors were re-installed at five WQIS1 through 5 on LCR and at one station on Kootenay River (WQ C2). 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 recorded data on 15-minute intervals. These sensors remained in place until the summer of 2012, when record high flows inundated the data logger component of the sensors and disabled them[1]. Replacement Onset® Water Level Logger (Model U20) pressure transducers were installed at each of the stations, except Kootenay River (WQ C2)[2], during the week of August 15 -18, 2012. The Onset logger recorded water levels every 20 minutes, but also required a barologger (Model U20) to compensate for

[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.

changes in barometric pressure and to measure air temperature. One barologger was installed at the top end of LCR in Reach 1 and another was installed adjacent to WQIS4 within the upland forest canopy. All pressure readings were compensated for barometric pressure and converted to water depth using HOBOware® software. Water depth was converted to elevation based on the length of the sensor cable and the surveyed elevation of the top of the stilling well.

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

During the seven years that the level loggers were recording data, individual levels loggers began to fail because of low batteries and/or malfunctions. At the time of documented failure, the sensors were removed from the river and sent back to the manufacturer for repair. The repaired sensors were then replaced during the next field session. This contributed to data gaps, especially in the latter half of the study.

8.3 Dataset

Table A7. Datasets used in the analysis of physical habitat management question #1.

Name/Description	Data Source	Years Obtained
LCR / Kootenay River Elevation / Water Temperature	Data collected at 5 stations (LCR) and 1 station (Kootenay River)	LCR - 2008 – 2018[1] Kootenay – 2011 – 2018[1]
Mean Daily Discharge at Hugh L. Keenleyside (HLK), Brilliant Dam (BRD), and at Birchbank (BIR)	Data obtained from Poisson Consulting	2008 - 2018
Kootenay Lake Temperature	Data obtained from FLNRORD. Water temperature data for Station KL8.	2008 - 2018[2]
Brilliant Dam Head Pond Elevation	Data obtained from Columbia Power Corporation	2008 - 2018
Castlegar Air Temperature	Data downloaded from Environment Canada	2008 - 2018
Arrow Lake Reservoir Daily Temperature	Data obtained from Poisson Consulting	2008 - 2018
Arrow Lakes Reservoir Hourly Elevation at Nakusp	Data obtained from Poisson Consulting	2008 - 2018

[1]Datasets are partial due to level logger malfunction, lack of data collection during contract transitions and level logger exposures during extremely low flow periods.
[2] Temperature data from Kootenay Lake were only available for one to two days in each season. A full temperature dataset was created by predicting daily water temperature from a Generalized Additive Model (GAM) of daily water temperature.

8.4 Analysis

To understand the diel scale of water temperature changes in the LCR the daily diel temperature range was calculated using tidbit data and a time series decomposition was conducted. The daily diel temperature range (DTR) was calculated from hourly tidbit data and was the difference of the maximum and minimum daily temperatures (Woolway et al. 2016). The DTR was only calculated for MD plates and was visually compared by site, year and season.

To understand if flows are affecting the water temperature of LCR, Random Forest (RF) models were used. A separate RF model was run for the three flow periods of interest MWF, RBT and FFF. Random Forest is a non-parametric machine learning technique which does not require random distribution of residuals and can accommodate categorical predictor variables (Read et al. 2015). RF is an appropriate technique for time series data because it can accommodate non-stationarity and autocorrelation (Naing and Htike, 2015). The explanatory variables used for RF models included reservoir temperature (Kootenay Lake and Arrow Lakes Reservoir), Castlegar air temperature, reservoir elevation (BRD head pond and Arrow Lakes Reservoir), mean daily discharge from HLK and BRD dams, and year. The analysis was independently undertaken for two sites (WQIS1 and WQIS5) to control for pseudo-replication among sites. WQIS1 was selected for modelling because it is upstream of the Kootenay River confluence, like WQIS2 and WQIS3, and was only influenced by HLK dam and Arrow Lakes Reservoir. The WQIS5 site was also selected for modelling because it is downstream of Kootenay River and was influenced by BRD and HLK dams, and Kootenay Lake and Arrow Lakes Reservoir.

Random Forest determines the importance of each predictor variable and the relationships between each predictor variable and response variable. The variable importance measure for each predictor is determined by calculating the mean decrease in prediction error (Mean Squared Error) if the predictor is dropped from the model (Liaw and Wiener, 2002). Predictor variables that have a strong relationship with LCR water temperature should have large variable importance. Dropping these predictors from the model causes a large increase in prediction error. Variable importance plots for all predictors included in each model were generated to help identify predictors associated with the LCR water temperature variables. Partial dependence plots were generated to better understand the relationship between the selected top predictor and the response variable while considering the effects of the other variables in the Random Forest model (Liaw and Wiener, 2002).

Random Forest uses Classification and Regression Tree (CART) models as the base model. CART is a non-parametric tree-based method that splits data into separate groups based on the response variable (De'ath and Fabricius 2000; Jun 2013). CART initially partitions the data into two groups based on a split point and splitting variable that minimizes the sum of squares of the response variable of each group (De'ath and Fabricius 2000; Hastie et al. 2001). A recursive algorithm is used to search through every possible combination of explanatory variables and values to determine the best splitting variable and split point (Hastie et al. 2001). The CART algorithm continues to make binary splits at each tree node until a stopping criterion is reached (Jun 2013).

Random Forest builds different CART models by bagging, using a subset, the data and the explanatory variables tried - at each split. Each CART model uses a random subset of the dataset and at each split in the tree a random subset of predictor variables is tried as a potential splitting variable (Jones and Linder, 2015). The default setting used in the R package Random Forest were used for the LCR water temperature models. The Random Forest models contain 500 trees

(CART models) and in our case, one of the predictor variables out of the five predictors was randomly chosen as the splitting variable at each node (Liaw and Wiener, 2002).

WQIS1 occurs above the confluence of the Kootenay River and only experiences flows from HLK, whereas WQIS5 occurs downstream of the Kootenay River confluence and is subject to flows from both HLK and BRD. To account for this, explanatory variables were standardized based on location. Flows from HLK and Arrow Lake Reservoir temperature and water elevation were used for WQIS1, while BRD /BBK flows were used for WQIS5.

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

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

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

Likewise, reservoir elevation was calculated using the following equation:

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

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

8.5 Results

8.5.1 Daily Patterns of Water Temperature

The daily diel temperature range (DTR) is the difference between the maximum and minimum daily water temperature. It was determined at moderately deep (MD) productivity plates during each deployment season. The daily temperature fluctuations were largest during the summer deployment and much smaller during the winter deployment (Figure A1). Daily winter temperatures fluctuated by less than 0.25 °C, while summer temperatures approached daily fluctuations of 0.5 °C. The daily fall fluctuations fell between summer and winter. Temperature fluctuations were generally consistent between productivity sites and across years.

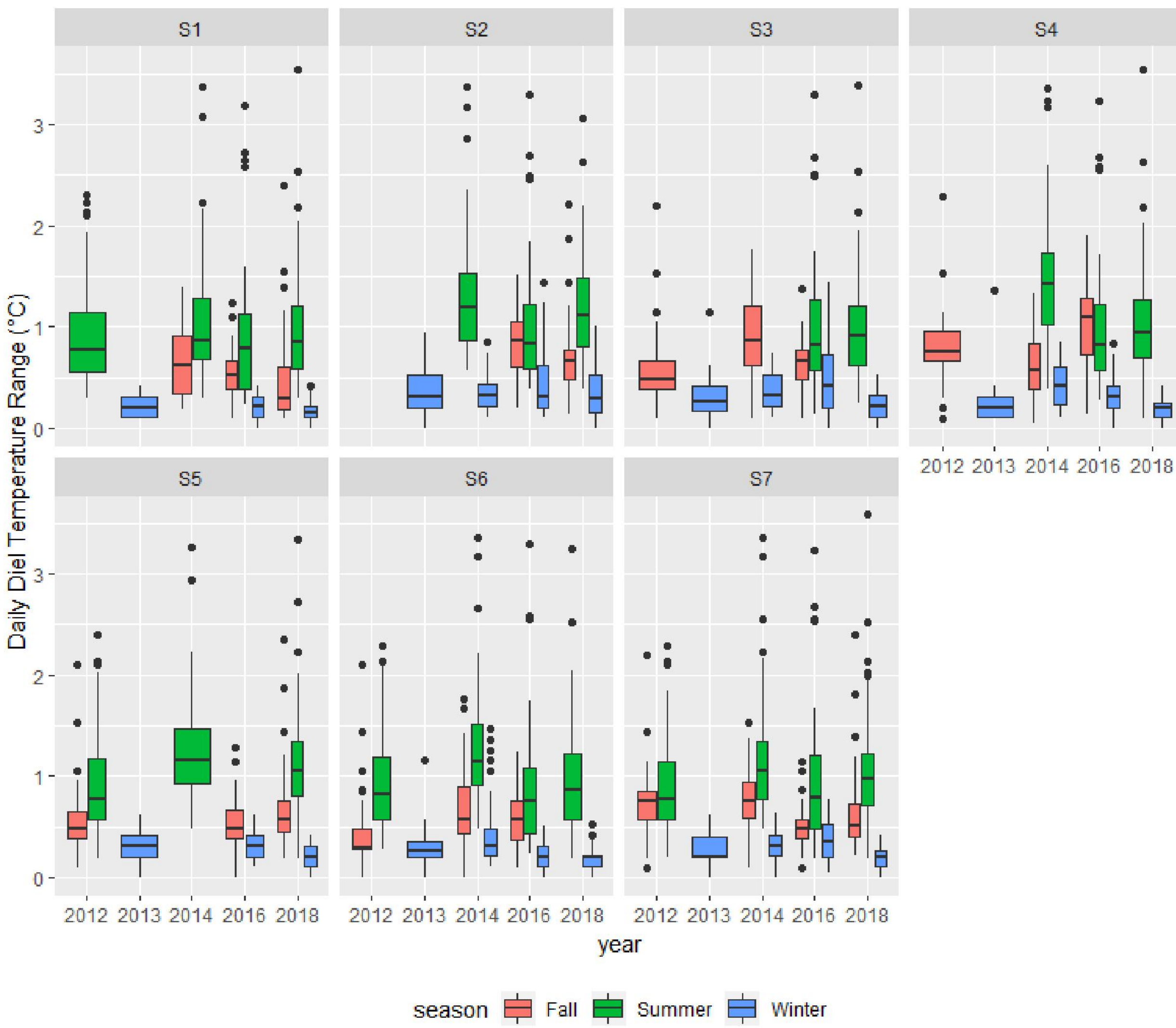

Figure A1. Daily Diel Temperature Range (DTR) for MD plates by year, season and site.

8.5.2 Effects of Flow on Water Temperature at WQIS1

To test the specific hypotheses that implementation of different flow periods may affect LCR water temperature, we ranked the relative importance of flow regime with other parameters that can affect water temperature including reservoir temperature, reservoir elevation, year and air temperature. The analysis was undertaken for two sites, WQIS1, which is upstream of Kootenay River, and WQIS5, which is downstream of Kootenay River. At WQIS1, LCR water temperature was most strongly correlated with Arrow Lakes reservoir temperature during the MWF flow period, and Arrow Lakes reservoir elevation during the RBT and FFF periods (Figure 4-9).

The five predictors explained 88-96% of the variation in LCR water temperature at WQIS1 (Table A8). The importance of each predictor varied during the different flow periods. Although flow played a role, it was never a critical predictor of LCR water temperature at WQIS1 (Figure A2, Figure A3 and Figure A4).

Table A8. Summary of LCR Water Temperature Random Forest Models at WQIS1 (mse=Mean Squared Error, rsq=model R^2).

mse	flow_period	rsq
0.020	MWF	0.960
0.820	RBT	0.930
0.630	FFF	0.880

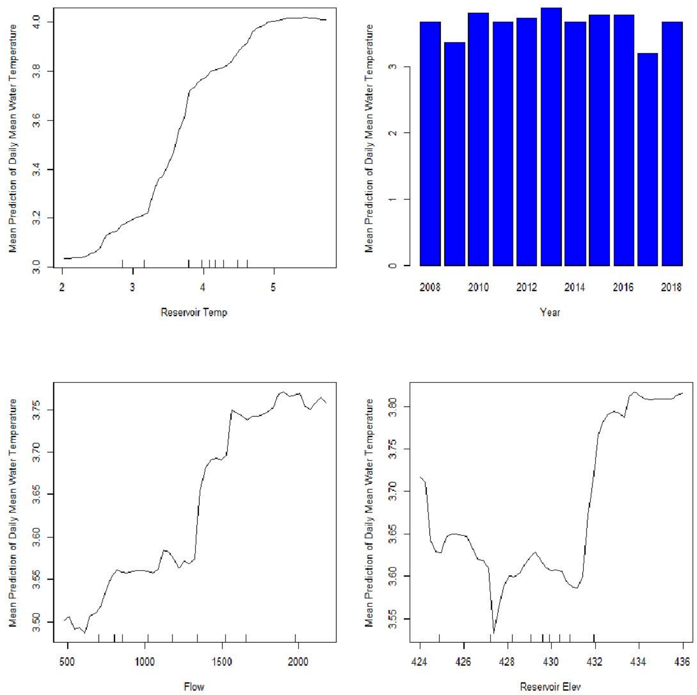

Figure A2. Random Forest Model partial dependence plots for the top four explanatory variables for Daily Mean Water Temperature at WQIS1 during the MWF flow period.

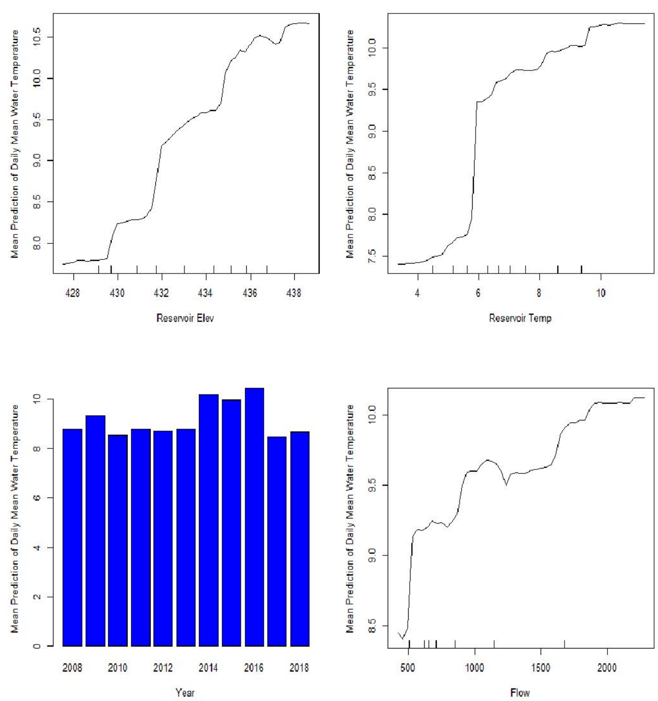

Figure A3. Random Forest Model partial dependence plots for the top four explanatory variables for Daily Mean Water Temperature at WQIS1 during the RBT flow period.

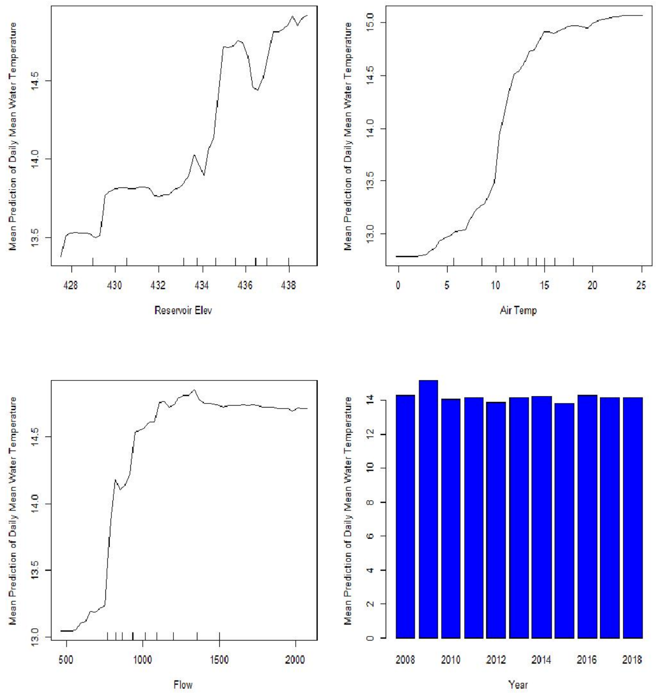

Figure A4. Random Forest Model partial dependence plots for the top four explanatory variables for Daily Mean Water Temperature at WQIS1 during the FFF period.

8.5.3 Effects of Flow on Water Temperature at WQIS5

This analysis of flow effect on water temperature was also undertaken at WQIS5, the furthest downstream site that is also influenced by the Kootenay River. At WQIS5, LCR water temperature was most strongly correlated with reservoir temperature during the MWF flow period, and flow during the RBT and FFF periods (Figure A5). Interestingly, flow was also the second most important predictor during the MWF flow period.

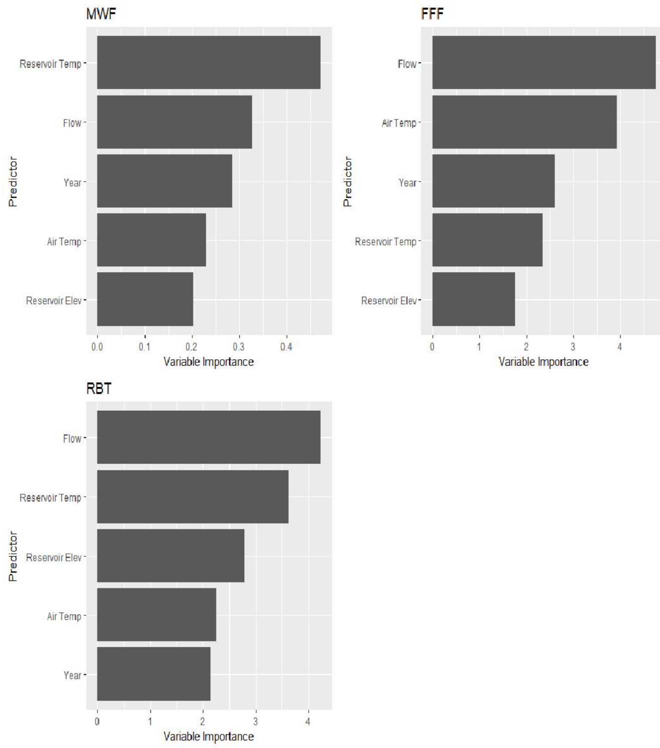

Figure A5. Random Forest Model variable importance plots for LCR water temperature during MWF, RBT and the FFF periods at WQIS5.

The five predictors at WQIS5 also explained a high percentage of the variation in LCR water temperature (80 – 94%) (Table A9). The importance of each predictor again varied during the different flow periods, but flow played a much more important role on water temperature at WQIS5 compared to WQIS1 (Figure A6, Figure A7 and Figure A8).

Table A9. Summary of LCR Water Temperature Random Forest Models at WQIS5 (mse=Mean Squared Error, rsq=model R2).

mse	flow_period	rsq
0.160	MWF	0.800
0.520	RBT	0.940
0.660	FFF	0.920

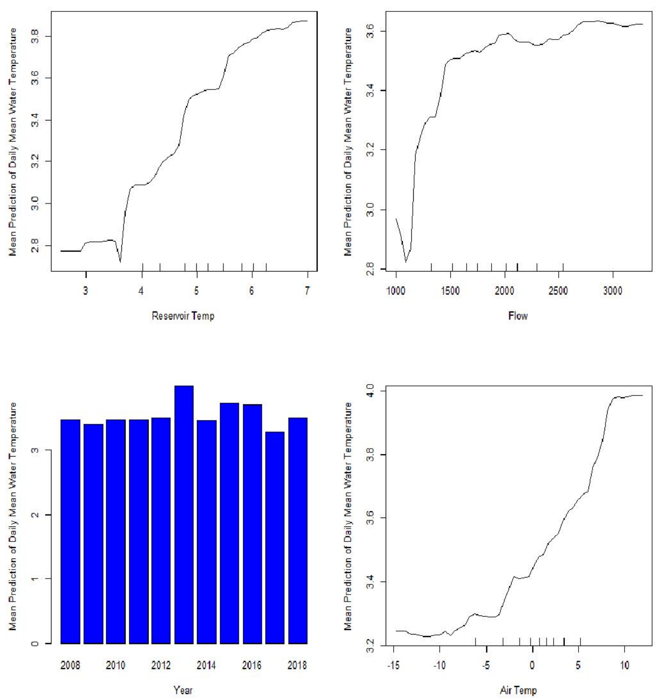

Figure A6. Random Forest Model partial dependence plots for the top four explanatory variables for Daily Mean Water Temperature at WQIS5 during the MWF flow period.

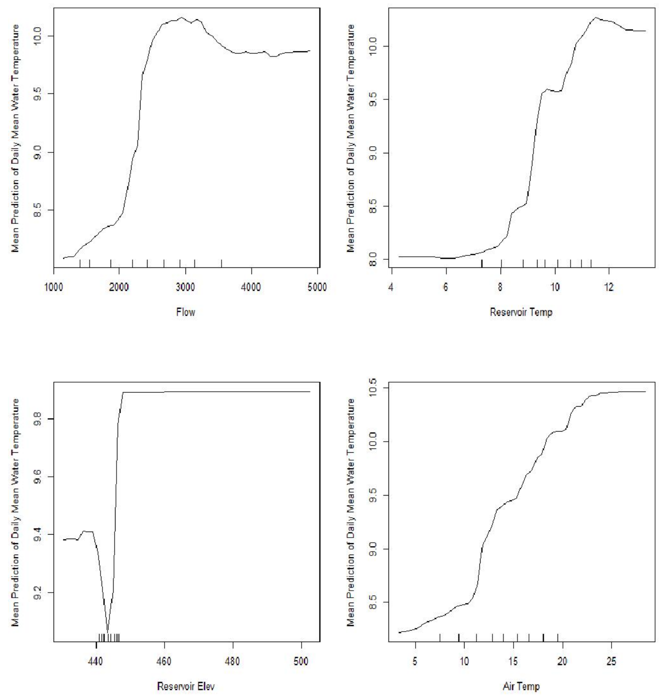

Figure A7. Random Forest Model partial dependence plots for the top four explanatory variables for Daily Mean Water Temperature at WQIS5 during the RBT flow period.

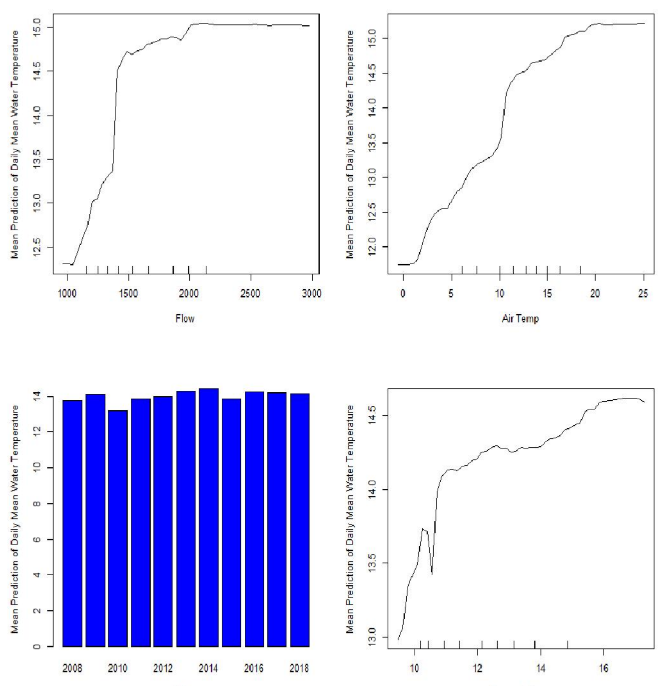

Figure A8. Random Forest Model partial dependence plots for the top four explanatory variables for Daily Mean Water Temperature at WQIS5 during the FFF period.

8.6 Discussion

Water temperature in LCR exhibits seasonally consistent patterns that are largely driven by seasonal changes. The greatest daily variability in water temperature occurs during the summer when temperatures may fluctuate by as much as a half degree Celsius. During the winter, water temperatures are consistently low, between 2.5 and 5°C, and there is very little daily fluctuation. In spring, the water temperatures begin to increase and ramp up rapidly to approximately 15°C by the end of June. The water temperatures continue to increase during the summer months, with temperatures above the Kootenay River confluence peaking at about 18°C, and temperatures below the confluence peaking closer to 20°C. As one would expect, the temperatures begin to decline in the fall and reach approximately 10°C by the end of October.

To understand the influence of flow on LCR water temperature, the RF model analyses were independently undertaken at two WQIS on LCR. WQIS1 is located approximately 1.8 km downstream of HLK Dam and thus is only influenced by flows originating from Arrow Lakes Reservoir. WQIS5 is located about 15 km downstream of the Kootenay River confluence, where flows from Kootenay River and LCR are well mixed.

The predictors of LCR water temperature included reservoir temperature, reservoir elevation, air temperature, year and flow. The importance of these predictors varied during the three different flow periods, but they explained 80-96% of the variation in LCR water temperature. Flow was a much more important predictor of water temperature at WQIS5 than at WQIS1. During the RBT and FFF periods, flow was the top predictor of water temperatures at WQIS5, whereas reservoir elevation was the top predictor of water temperatures at WQIS1.

Flow played a larger role on influencing water temperature at sites downstream of Kootenay River because Kootenay River tends to be slightly warmer than LCR water temperatures. Depending on the flows originating from Brilliant Dam, the warmer temperatures can strongly influence LCR water temperatures downstream of the confluence. Slightly warmer temperatures at WQIS4 and 5 were consistently observed, with summer temperatures peaking almost 2°C higher than upstream sites.

These findings suggest that flow does influence LCR water temperature, but its effects are small compared to reservoir elevation, reservoir temperature and air temperature at sites upstream of the Kootenay River. At downstream sites, flow is a much more important predictor of water temperature, but it is largely due to the influence of Kootenay River flows which are slightly warmer than LCR flows, and not due to the managed MWF, RBT or FFF regimes from HLK Dam. We therefore accept the null hypothesis Ho_{1phy} which states that continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall, do not alter the seasonal water temperatures regime of LCR.

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9.0 APPENDIX 4. Physical Habitat - Management Question #2

9.1 Introduction

This appendix addresses physical habitat management question #2 and associated hypotheses.

MQ#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?*

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.*

9.2 Methods

Water elevation field data collection methods were previously presented in the methods section of Appendix 3 because they are consistent with the water temperature data collection. In short, data was obtained from WQIS1-5 on LCR and from a single location on Kootenay River (WQ C2). River elevation data collection began in 2008 and continues in 2019, however datasets have missing data due to causes including level logger malfunction, lack of data collection during contract transitions and level logger exposures during extreme low flow periods.

9.3 Datasets

Table A10. Datasets used in the analysis of physical habitat management question #2.

Name/Description	Data Source	Years Obtained
LCR / Kootenay River Elevation / Water Temperature	Data collected at 5 stations (LCR) and 1 station (Kootenay River)	LCR - 2008 – 2018* Kootenay – 2011 – 2018*
Mean Daily Discharge at Hugh L. Keenleyside (HLK), Brilliant Dam (BRD), and at Birchbank (BIR)	Data obtained from Poisson Consulting	2008 - 2018

*datasets are partial due to several reasons including level logger malfunction, lack of data collection during contract transitions and level logger exposures during extremely low flow periods.

9.4 Analysis

The mean 2018 water level elevations recorded at WQIS1-5 in LCR and WQ C2 in Kootenay River were compared to the combined water elevation (± SD) during all years. Subsequent analysis of the effects of water level during MWF and RBT flow periods relied on the following key assumptions:

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

9.4.1 Mountain Whitefish Flow Period

To address the sub-hypothesis H_{O2Aphy}, that states continued implementation of MWF flows does not reduce the river level difference between the maximum peak spawning flow (Jan 1 to Jan 21) and the minimum incubation flow (Jan 21 to Mar 31), the water elevation difference between the maximum elevation during spawning and minimum elevation observed during incubation at each WQIS was investigated. Because historic river elevation data was not available, predicted elevations were calculated from flow data. The methods used in this analysis were the same as those reported for previous years (Olson-Russello *et al.* 2015; Plewes et al. 2017). These methods used the whole annual dataset rather than a subset of the flow period to increase the accuracy of the predicted elevations. The predicted elevations were then subsequently subset by flow period for further use in the analysis. Candidate linear regression models of water elevation were constructed for each WQIS, containing all combinations of flows from HLK, BRD, and BBK, and their associated quadratic terms (flow values^2) as explanatory variables (Table A11). Quadratic terms and appropriate data transformations were considered to account for potential logarithmic or non-linear relationships between flow and elevation. Model selection via Akaike information criterion corrected for small sample sizes (AICc) was used to determine the best fit and other plausible models (Δ AICc<3). In this approach, candidate models were considered and ranked based on their AICc scores. The best fit model exhibited a trade-off between model complexity and optimal fit of regression.

The top model for each site was then used to predict water elevation for periods between pre-implementation of MWF flows (1984 to 1994), post-implementation of MWF flows (1995 to 2007), and continuation of MWF flows (2008-2018). Differences among predicted elevations during each flow period were tested using a permutation ANOVA and subsequent post-hoc analysis (Tukey's HSD) to determine groupings. The permutation ANOVA was used in lieu of traditional ANOVA or Student's t tests because it does not require the same assumptions of normality and was preferred to non-parametric methods due to ease of interpretation of results and the ability to conduct post-hoc analyses. Finally, the data were compared to actual elevations measured during 2008 - 2018 to investigate how predicted elevations compared to field collected elevations.

Table A11. Flow Combinations used in Regression Modeling for Predicting Water Levels during the MWF and RBT Flow Periods

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

9.4.2 Rainbow Trout Flow Period

To address sub-hypothesis HO_{2Bphy}, that states continued implementation of RBT flows does not maintain constant water level elevations at Norns Creek fan between April 1 and June 30, we used the same analysis procedure described above for sub-hypothesis HO_{2Aphy}. To limit the analysis to the Norns Creek fan, the closest two sites, WQIS2 and WQIS3, were included. To evaluate the cumulative elevation differences over the RBT flow period, linear regressions of water elevation were constructed for each site, containing all combinations of flows from HLK, BRD, and BBK, and their associated quadratic terms as explanatory variables (Table A11). The same model selection process was used to determine the best fit model of all plausible models (Δ AICc<3) and subsequently predict elevation during pre-implementation of RBT flows (1984-1991), implementation of RBT flows (1992-2007), and continued RBT flows (2008-2016). Differences among predicted elevations during each time period were again tested using a permutation ANOVA and subsequent post hoc analysis (Tukey's HSD) to determine groupings. Finally, the data were compared to actual elevations measured in 2008-2018 to investigate how predicted values compared to those collected in the field.

9.5 Results

9.5.1 Water Levels in 2018

The water levels in 2018 differed considerably from those of previous years. During the MWF flow period, mean daily water levels at LCR stations 1-5 were higher and more variable than typical. (Figure 4-10). The water levels then dropped so significantly on April 1 that the loggers were exposed and unable to record data until late April when the levels began to increase drastically and peaked in late May. This was unusual as the peak water levels have typically occurred in early July (Olson-Russello *et al.* 2015; Olson-Russello *et al.* 2014; Olson-Russello *et al.* 2012; Plewes *et al.* 2017).

For most of the RBT flow period, mean daily water levels at stations 1-5 were well above normal. During the FFF, stations 1-3 exhibited higher variability and lower levels near the end of the flow period. The low flow levels in October were smaller at the downstream stations 4 and 5 relative to the upstream stations (Figure 4-10).

The Kootenay station (WQ_C2) had more typical water levels during the MWF period, followed by much lower water levels than previous years during the RBT flow period. The mean daily water level of WQ_C2 during the FFF period was higher than previous years (Figure 4-10). In 2018, successfully recorded water level elevations above the Kootenay River confluence ranged from ~417.6 to 422.1 m asl. Below the confluence (WQIS4 and 5), elevations ranged from ~410.6 to 418.1 m asl. The maximum mean daily river flow recorded in 2018 were 4473.9 m^3/s on May 26th. For comparison, flows recorded in previous years of this study (2011, 2012, 2013, 2014 and 2016) were 4,155.4 m^3/s on July 9th; 6,043.1 m^3/s on July 21th; 4,434.4 m^3/s on July 5th; 3,677.9 m^3/s on July 8th and 3,142.7 m^3/s on June 14th.

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

The following results address Ho_{2Aphy} which investigates the influence of MWF fish flows on river water levels. All relationships between flow and river water levels were statistically significant ($p < 0.05$). At all the WQ sites, the predicted elevation difference during pre-MWF flows (1984-1994) was significantly higher than the predicted elevation difference during post and continuous flow periods (permutation ANOVA, d.f. 3, $p<0.001$). The accuracy of the predictive elevations is supported statistically and by comparing field measured elevations to the predicted elevations during the post-implementation period (Figure A9).

Statistical analyses of the flow and water elevation data indicate that the implementation of MWF flows has been effective at reducing the difference between maximum flow during MWF spawning and minimum flow during MWF incubation and should benefit the fishery (Table A12). These results are consistent with findings by Scofield *et al.* (2011) and all other annual reports (i.e. Olson-Russello *et al.* 2015; Plewes et al. 2017).

The best models varied among the five WQIS sites and contained different sets of explanatory flow variables. The variance in elevation described by top models was typically very high (R^2 range: 0.90-0.98), suggesting that the use of these models for predictive purposes is plausible (Table A12). The accuracy of the predictive elevations is further supported when the actual elevation differences during the post implementation period are compared to the observed elevations.

Table A12. The best fit models for each water quality index station that were used to predict historic water levels for the MWF and RBT flow periods

Site	Best Fit model (Intercept + Coefficient(±SE))	Adjusted R^2	p-Value
WQIS1	417.5 + BIR(0.000147 ± 4.81e-05) + BIR^2 (-3.15e-08 ± 1.12e-08) + BRD(-0.000269 ± 3.97e-05) + BRD^2(2.69e-07 ± 1.70e-08) + HLK(0.00256± 5.51e-05) + HLK^2(-3.12e-07 ± 2.12e-08)	0.948	< 0.001
WQIS2	417.2 + BIR(0.000314 ± 2.91e-05) + BIR^2 (-4.36e-08 ± 6.30e-09) + BRD(-0.000302 ± 2.62e-05) + B RD^2(2.70e-07 ± 1.04e-08) + HLK(0.00278± 3.62e -05) + HLK^2(-4.11e-07 ± 1.38e-08)	0.974	< 0.001
WQIS3	416.4 + BIR(0.000365 ± 2.56e-05) + BIR^2(-4.21e-08 ± 5.58e-09) + BRD (-0.00014 ± 2.31e-05) + BR D^2(2.98e-07 ± 9.25e-09) + HLK(0.00194 ± 3.15e-05) + HLK^2(-1.77e-07 ± 1.2 0e-08)	0.979	< 0.001
WQIS4	409.85 + BIR(0.00173 ± 8.30e-05) + BIR2 (-1.89e 07 ± 1.78e-08) + BRD(0.00088 ± 7.42e-05) + BRD2(4.77e-08 ± 2.91e-08) + HLK(0.00079± 9.92e-05) + HLK2(8.68e-08 ± 3.74e-08)	0.904	< 0.001
WQIS5	409.1+ BIR(0.00037 ± 3.29e-05) + BIR^2(-6.70e-08 ± 5.93e-09) + BRD(0.00117 ± 3.85e-05) + BRD^2(9.46e-08 ± 1.17e-08) + HLK(0.00102 ± 4.25e-05) + HLK^2(1.41e-07 ± 1.53e-08)	0.974	< 0.001

NOTE: BIR = Birchbank, BRD = Brilliant Dam, HLK = Hugh L. Keenleyside Dam, MWF = Mountain Whitefish flows, RBT = Rainbow Trout flows

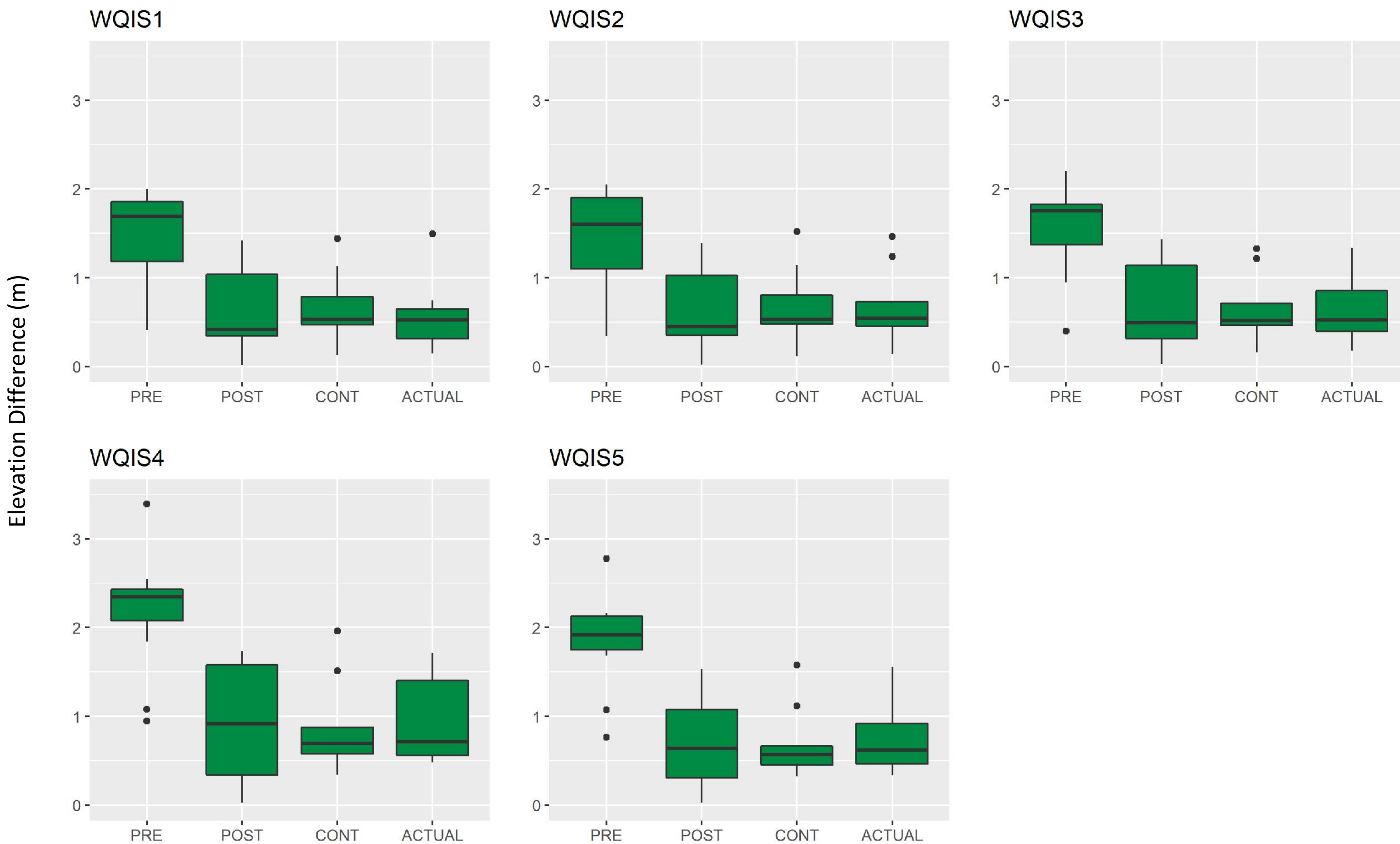

Figure A9. Predicted water level elevation difference between maximum flows during Mountain Whitefish (MWF) spawning (Jan 1 – Jan 21) and minimum flows during MWF egg incubation (Jan 22 – Mar 31) for Pre (1984 – 1994), Post (1995-2007), and Continuous (2008-2018) flow years at each water quality index station. The actual dataset is included to illustrate variability between the predicted continuous (CONT) values and actual elevation field data collected during the 2008-2018 study period.

9.5.3 Rainbow Trout Flow Period (HO_{2Bphy})

The following results address HO_{2Bphy}, which investigates the effects of RBT flows on water levels at Norn's Creek Fan between April 1 and June 30. The results are derived from analyses described in the previous section. The best statistical models for the sites WQIS2 and WQIS3 that are located near Norn's Creek Fan included BBK, BRD and HLK flows (Table A12). At both sites, flow had a strong positive effect on water elevation.

For both WQIS, the total elevation drop was significantly higher during pre-implementation of RBT flows (1984-1991) than during post (1992-2007) and continuous (2008-2018) flow periods (permutation ANOVA, d.f. 3, $p<0.001$,) (Figure A10). Like the results for MWF, RBT data shows good agreement between predicted and observed water elevations.

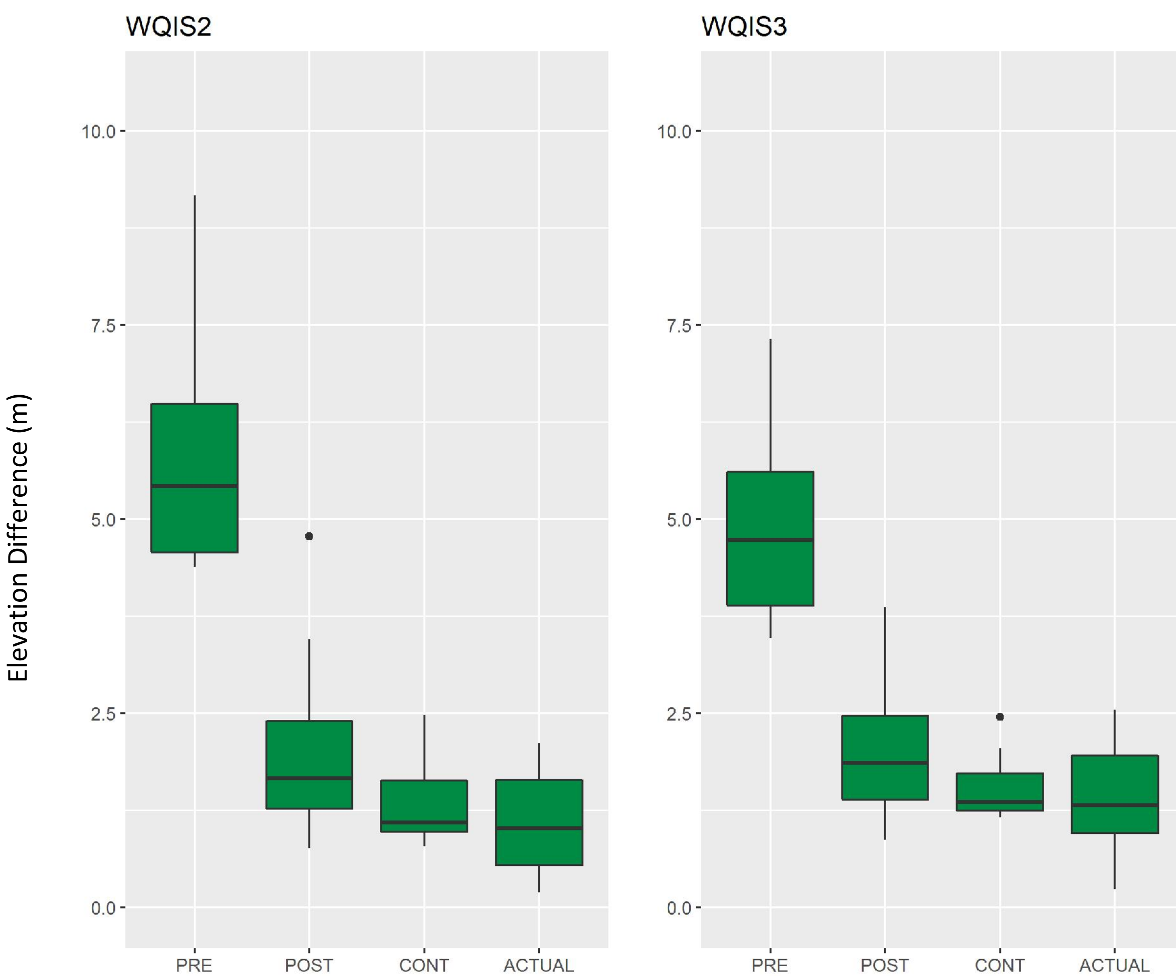

Figure A10. Cumulative sum of elevation drops occurring during the Rainbow Trout Flow period for Pre (1984 – 1991), Post (1992-2007), and Continuous (2008-2018) flow years at each water quality index station. The actual dataset is included to illustrate variability between predicted CONT values and actual elevation field data collected during 2008 -2018.

9.6 Discussion

The water levels in 2018 differed considerably from previous years. During the 2018 MWF flow period, mean daily water levels at LCR stations 1-5 were higher and more variable than usual. The water levels then dropped significantly on April 1 and loggers were exposed and unable to record data until late April when the water levels increased drastically and peaked in late May. This was unusual, as peak water levels typically peak in late June or early July.

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

The modeling data indicate that both post-implementation (1995 – 2007) and continued (2008 – 2018) MWF flow periods resulted in smaller changes in water elevation between the spawning and incubation periods than pre-implementation of the flow regime (1984 – 1994). We expect reasonable strength in this relationship because predicted elevations were not different from those measured in the field for the period assessed. We therefore reject the management sub-hypothesis HO_{2Aphy} and conclude that MWF flows have reduced the river level difference between the maximum peak spawning (1 Jan to 21 Jan) and the minimum incubation flow (21 Jan to 31 Mar).

During the RBT flow period, the modeling data for WQIS2 and WQIS3 indicate that both the post-implementation and the continued RBT flow regimes caused a smaller cumulative decrease in river elevation than before the flow regime was implemented. Like the MWF flow period analysis, modelled water elevations and those measured in the field were similar. We therefore reject management sub-hypothesis HO_{2Aphy} that RBT flows does not maintain constant water level elevations at the Norns Creek fan between 1 Apr and 30 Jun.

9.7 References

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10.0 Appendix 5. Physical Monitoring - Management Question #3

10.1 Introduction

This appendix addresses the Physical Management Question #3 and associated management hypotheses.

MQ#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?*

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.*

10.2 Methods

10.2.1 Data Collection

Chemical and physical water quality parameters were collected at seven sampling locations between 2012 - 2015 (Table 3-1). The number of water quality sampling locations was reduced from ten to seven, as per a recommendation put forth in Year 4 (2011) when flows in Blueberry, China and Champion Creeks were recorded as minimal to nil throughout several of the sampling sessions (Olson-Russello *et al.* 2012).

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

Since 2012, sampling was modified from the previously collected monthly samples in the June to October growing season to allow sampling to be more disbursed over the year and to achieve a greater overlap with the MWF flow period. For example, in 2015, sampling took place on April 1, June 24, August 18, and October 20, 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.

Conductivity, TDS, alkalinity and pH were sampled to address electrochemistry, while the nitrate, ammonia, and ortho-phosphate (SRP) analyses addressed the biologically active nutrients. Nutrients occurring as organic particulates require bacterial digestion before they are returned to a biologically active form, and of these, total phosphorus, TKN and total nitrogen were analyzed.

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

The sample bottles were provided by Caro Environmental Laboratories (Caro Labs) with the appropriate preservatives pre-measured into the bottles. The non-filtered samples were analyzed for total hardness, ammonia as nitrogen (N), nitrate as N, nitrite as N, total phosphorus, ortho-phosphorus, TDS, total suspended solids (TSS) and turbidity according to Standard Methods. Field-filtered samples were analyzed for low-level soluble reactive phosphorus (SRP) and total dissolved solids (TDS). TKN and SRP/ortho-P were also selectively sampled in certain years (e.g. 2009, 2013, 2014 and 2015). The filled sample bottles were placed on chipped ice and delivered to Caro Labs in Kelowna, BC within 24 hours of collection. One randomly chosen field duplicate and one deionized water travel blank were collected on each field trip. Additional QA/QC protocols were undertaken at Caro Labs.

10.3 Datasets

Table A13. Datasets used in the analysis of physical management question #3.

Name/Description	Source	Years Obtained
Water Quality Parameters (lab electrochemistry and nutrients) and Field Meter Parameters (temp. DO, Cond., TDS, pH)	Data collected at 5 stations on LCR, and as many as 5 tributary creeks.	(LCR + Norns Creek + Kootenay River 2008 – 2015, n = 36/site) (Blueberry/China/Champion creeks 2008 – 2011, n = <20/site)

10.4 Analysis

Water quality data from all years was combined for analysis (2008-2015). Data consisted of point samples from each WQIS collected four times annually and analyzed for approximately 15 parameters. If a measurement was non-detectable, it was entered into the database as ½ the lab reportable detection limit.

To illustrate the variation between season and year, boxplots of eight water quality parameters were generated using R. Sample number (n) at each LCR site and season ranged from 2 to 17 per season. Winter had the smallest number of sample sessions (n=3), spring and fall were intermediate (n=9 and 10) and summer had the greatest (n=19). To control for the effect of intraseasonal variability some water quality data was exclude from linear mixed effects models. For the summer linear mixed effects models only the water quality data that was collected in August was included. There were two different sampling events in June of 2010, only the June 30, 2010 sampling event was included in the spring linear mixed effects models.

The tested LCR water quality responses of interest included: conductivity, total dissolved solids, total phosphorus as phosphorus and nitrate + nitrite. Linear mixed effects models were used to determine if there were annual differences in the electrochemistry and nutrients of LCR during the MWF, RBT and FFF flow periods. To determine annual differences, a linear mixed effects model was run separately for each flow period. The level of pseudo-replication for these models is expected to be site. Site explained ~0% for some of the models, for these water quality models a regression was run. Regressions were run for turbidity for winter, summer and fall; winter pH, total phosphorus as phosphorus, and nitrate + nitrite; and fall nitrate + nitrite. The 95% confidence intervals for the fixed coefficient of year were calculated and plotted using the R package jtools version 2.0.1 (Long 2019).

10.5 Results

10.5.1 pH (field meter)

Over the years, pH values have occasionally exceeded the LCR upper pH objective limit of 8.5, but they remained below the BC MOE guideline for aquatic life of 9.0. (Figure A11). Since this is field meter pH, calibration can drift. pH at mainstem LCR sites averaged 7.8 ± 0.53 and ranged from 6.7 – 8.9, with the lowest values recorded in summers. For reference, pH in the Arrow Lakes Reservoir upstream of the HLK dam was less variable at 7.87 – 7.98 in Apr-Nov 2014 (BC MoE data).

pH in the LCR mainstem was stable, with a small increase in maximum pH and in pH variability in summer. The growing seasons with low flow regimes (summer, fall) had more variable pH than spring and winter. The lower pH objective of 6.5 has not been exceeded in the LCR during this study.

pH increased below the confluence with Kootenay River in every season. Winter pH at the sites above the Kootenay confluence averaged 7.8 and were similar to pH measured in other seasons. Photosynthesis raises pH and increased summer pH in the Kootenay River from an overall average of 7.8 in spring to an average of 8.2 in summer. Throughout the study period, Kootenay River showed the narrowest pH range of all the sample sites at 7.2 – 8.5, while Norns Creek exhibited the widest range of pH at 6.0 – 8.6.

A small pH decline pH from 2008 through 2015 may be occurring and was most evident in the winter, summer and fall models (Figure A12).

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

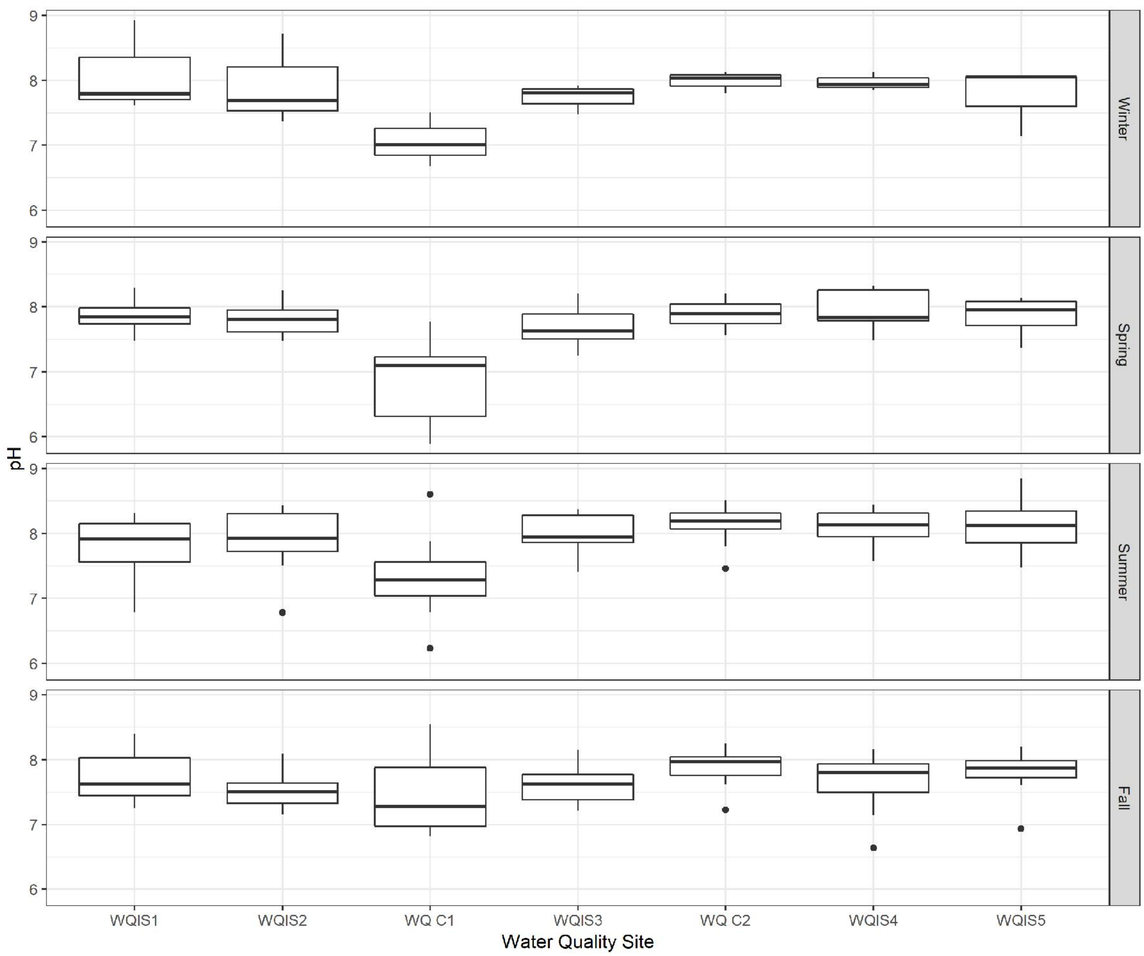

Figure A11. pH from LCR Water Quality Index Sites and Main Tributaries (2008-2015). The LCR lower and upper pH objective limits are 6.5 and 8.5, respectively, and the maximum BC MOE guideline for the protection of aquatic life is 9.0.

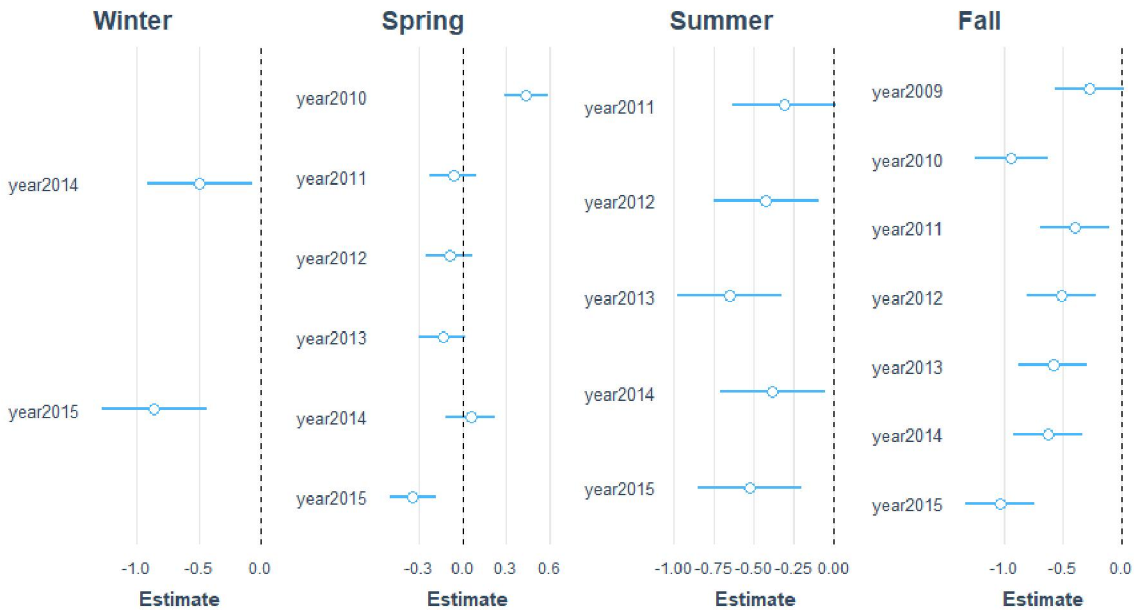

Figure A12. Fixed effects coefficients with 95% confidence intervals for year comparing pH from LCR Water Quality Index Sites. The reference year for winter=2013, spring=2009, summer=2008 and fall=2008.

10.5.2 Electrochemistry Parameters

Specific conductance, total dissolved solids (TDS), alkalinity and hardness all measure the concentrations of ionized constituents in water and they frequently trend together (Table A14). There is some overlap in the measured ions. For example, hardness and conductivity both include calcium. Conductivity and TDS were measured by field meter at every site on all trips. TDS was also analyzed at Caro Labs, while selected samples were submitted for alkalinity and hardness analyses.

Table A14. Ions Contributing to Electrochemistry Parameters

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

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

Specific conductance was monitored using a field meter. Historically in both LCR and its tributaries, specific conductance showed an inverse relationship with flow. On average, spring freshet of moderate flow years such as 2013 and 2014 had higher conductivity readings than in high freshet years 2011 and particularly 2012, that had significantly lower conductivity (Figure A15). Similarly, years with lower dam releases reduced dilution of base flows that include groundwater and resulted in higher conductivity throughout the LCR mainstem (Figure A14). Average conductivity at the downstream WQIS5 site measured in this study were within the range of specific conductance measured at Birchbank between 1983 and 1996 (105 – 160 µS/cm) (Holmes and Pommen 1999).

Throughout the study, Kootenay River had higher specific conductance measurements in all seasons compared to LCR (Figure A14). Norns Creek values were low, averaging <60 µS/cm in all seasons, consistent with historic values. Like the LCR, Norns Creek conductivity was highest in the three winter low flow sampling sessions, and lowest in spring freshet samples.

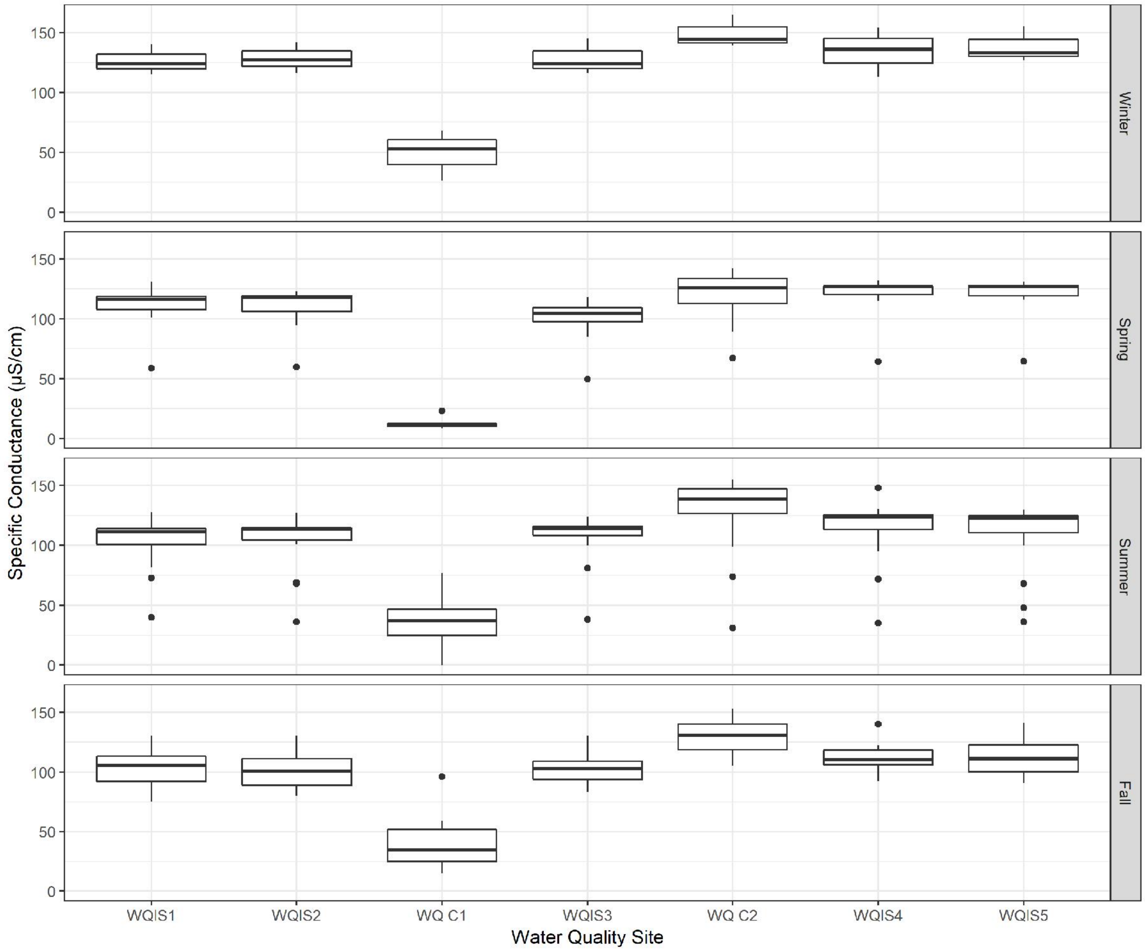

Figure A14. Conductivity from LCR Water Quality Index Sites and Main Tributaries (2008-2015). No guideline or objective is set.

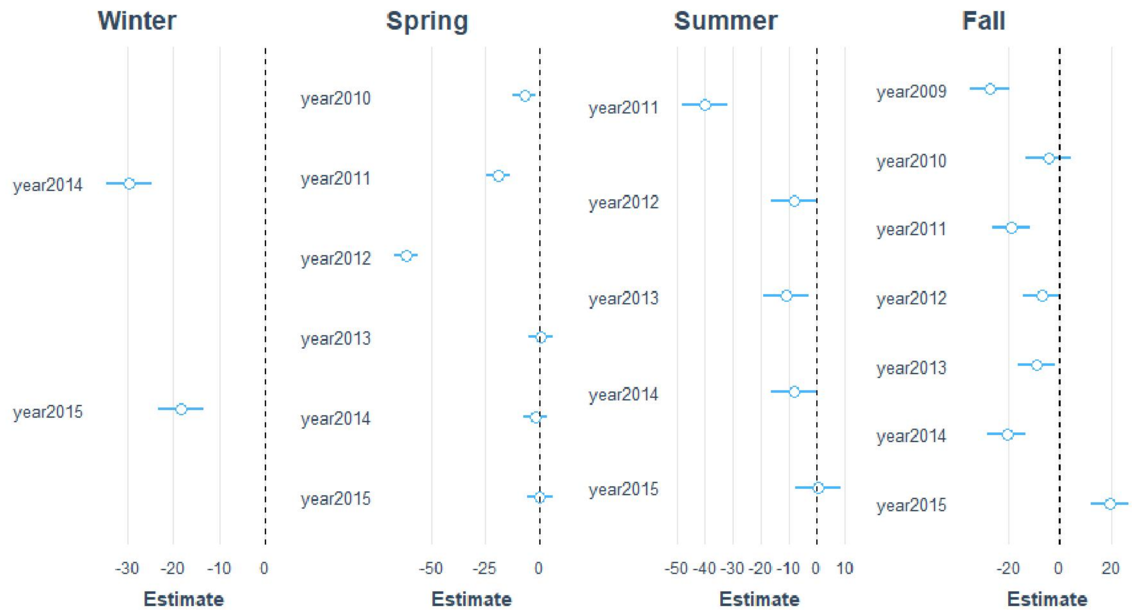

Figure A15. Fixed effects coefficients with 95% confidence intervals for year comparing conductivity from LCR Water Quality Index Sites. The reference year for winter=2013, spring=2009, summer=2008 and fall=2008.

Total dissolved solids are summarized in Figure A16. TDS showed similar patterns to field-measured specific conductance throughout this study. At mainstem sites, mean TDS was highest in winter (73 mg/L) and lowest in high flow periods (65 mg/L). In some years, elevated TDS occurred in spring because sampling occurred a month ahead of peak freshet flows. TDS tended to increase as water travelled through LCR and that increase was most evident in the fall and winter low flow periods.

TDS in Kootenay River usually exceeded that of LCR in all seasons and years of study. The higher TDS in Kootenay River (winter 88 mg/L; spring 70 mg/L) was reflected in observed increases in TDS at LCR sites downstream of their confluence in all seasons but winter. This was particularly evident at WQIS4 during the summer and fall seasons (Figure A16).

Norns Creek had consistently lower conductivity and TDS (winter 36 mg/L; spring 14 mg/L) than the mainstem sites, even during very low flow periods such as fall and winter.

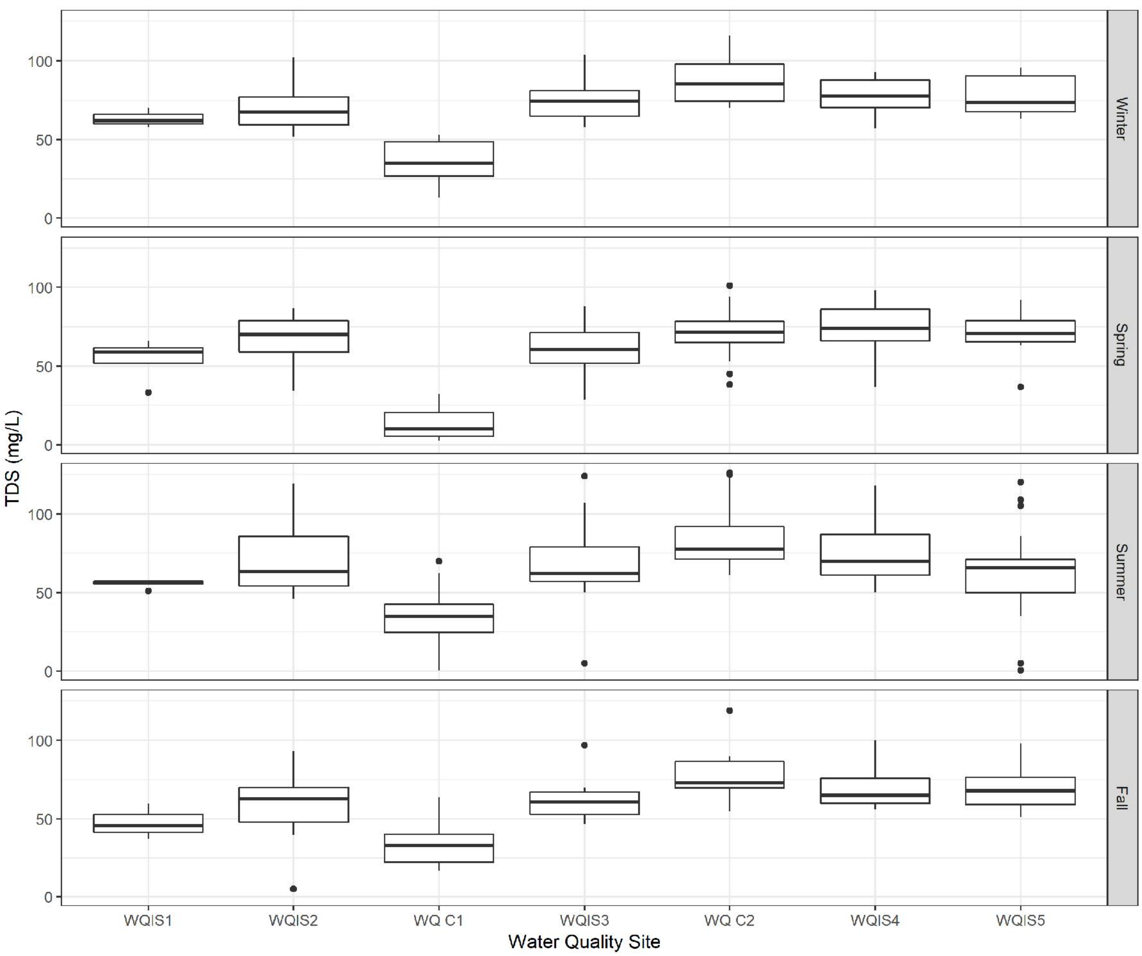

Figure A16. Total Dissolved Solids from LCR Water Quality Index Sites and Main Tributaries (2008-2015). No guideline or objective available.

10.5.3 Inorganic Nitrogen

The forms of inorganic nitrogen include nitrate, ammonia and nitrite and these are important macronutrients that are repeatedly consumed, transformed and released as water travels downstream. The ALR is the primary inflow to LCR and donated 0.062 - 0.177 mg/L as N (upper 20 m spill and release) during Apr-Nov 2014 (BC MoE). All LCR sites, Kootenay River and Norn's Creek were far below the BCMOE aquatic life nitrogen guidelines of 3 mg/L nitrate and 0.7 mg/L ammonia. Ammonia and nitrite were consistently non-detectable (<0.02 and <0.01 mg/L, respectively) in all years, as is expected in aerobic riverine environments.

As is often the case in rivers, inorganic nitrogen is dominated by nitrate throughout the LCR. Nitrate concentrations in LCR mainstem winter samples remained above 0.10 mg/L as N (0.119 – 0.149 mg/L), while samples from the balance of the year averaged concentrations below 0.10 mg/L as N. The highest concentrations of inorganic nitrogen occurred in the winter low flow period. Winter 2013 MWF flows were similar to unregulated flows and nitrate concentrations were significantly higher than those of 2014 and 2015. Spring nitrate concentrations were significantly higher in the high freshet years (2011, 2012) (Figure A18). Similarly, summer nitrate concentrations were highest in 2012 – a high flow year. These results suggest that nitrate concentrations increase with flows.

Nitrate concentrations in the LCR were elevated in the fall at sites closest to the HLK dam, possibly as a result of the fertilization program on the Arrow Lakes Reservoir (Larratt *et al.* 2013) (Figure A17). Nitrate (DIN) at WQIS1 was positively correlated with DIN at AR-8, ($r=0.56$, $p=0.005$).

Like ALR, a fertilization program is also active on Kootenay Lake. In winter and spring, Kootenay flows had more nitrate than LCR, while in summer and fall, the reverse was true. In the high flow years of 2011 and 2012, Kootenay River had similar nitrate concentrations to LCR during freshet (spring) but declined during the clear flow period (summer and fall). In 2013 and 2014 with moderate freshets, the spring concentrations were elevated to 0.075 and 0.105 mg/L NO_3 as N, respectively.

Nitrate concentrations were much lower in Norns Creek than at the mainstem sites and averaged 0.015 ± 0.04 mg/L NO_3 as N over the course of this study. Nitrate was highest in winter low flows. For the balance of the year, Norns Creek had consistently low nitrates but moderate phosphorus concentrations.

The US EPA found that Columbia River's nitrate load increased to almost twice its historical loads during the latter half of the 1990s, but by 2014, the nitrate load had returned to levels slightly greater than those seen in the late 1970s (US EPA 2016).

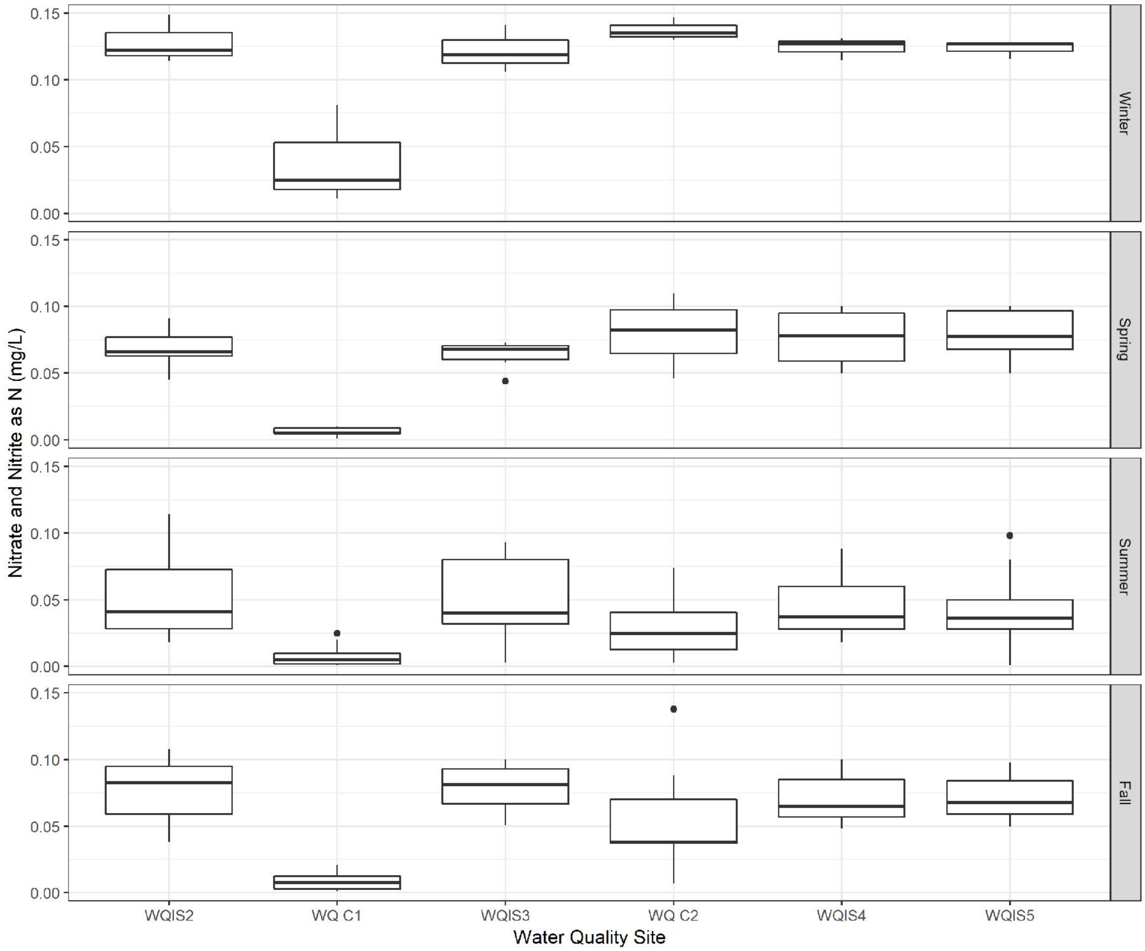

Figure A17. Nitrate and Nitrite from LCR Water Quality Index Sites and Main Tributaries (2008-2015). BC MOE guideline for the protection of aquatic life is 3 mg/L nitrate; 0.7 mg/L ammonia.

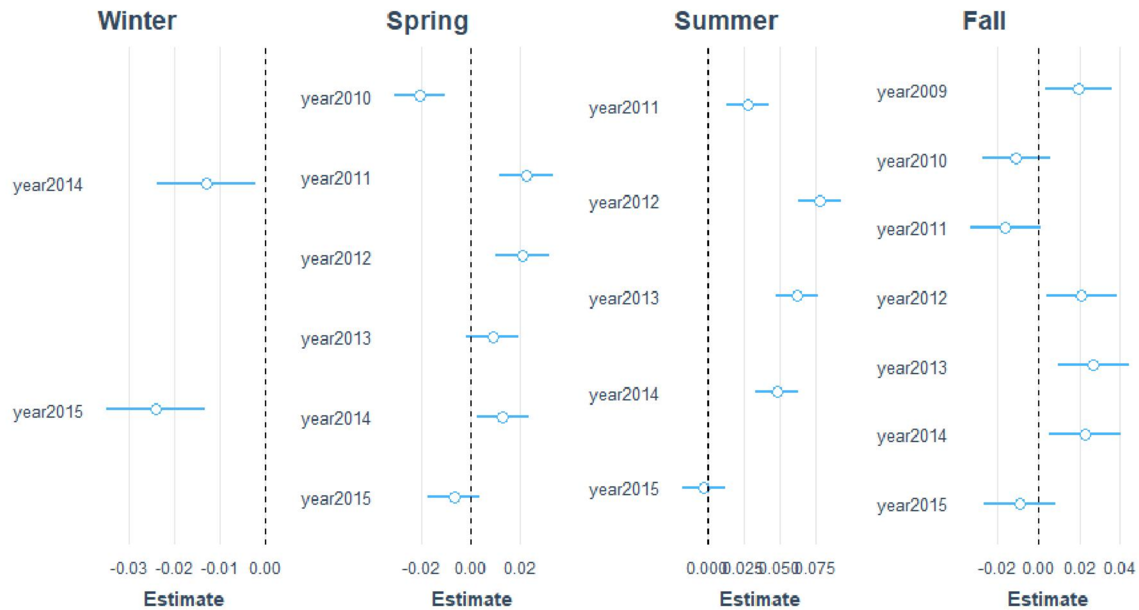

Figure A18. Fixed effects coefficients with 95% confidence intervals for year comparing nitrate and nitrite from LCR Water Quality Index Sites. The reference year for winter=2013, spring=2009, summer=2008 and fall=2008.

10.5.4 Organic Nitrogen and Total Nitrogen

Total Kjeldahl nitrogen (TKN) measures ammonia and organic forms of nitrogen which include N in algae, detritus, woody debris, etc. Since ammonia was consistently non-detectable in all LCR data, TKN can be assumed to represent organic nitrogen. Significant N sources above LCR include the ALR fertilization program, while municipal effluent, mill operations and N-enriched groundwater can also be significant contributors directly to the LCR. These sources can be organic N, or be converted to organic N by periphyton. No Objectives for organic or total N have been set for the Columbia River (Butcher, 1992) but a typical large river total N budget would range from 0.15 – 0.5 mg/L with a large TKN component (Wetzel, 2002). TKN samples were only collected in 2009, 2014 and 2015, thus a box plot could not be made.

The LCR mainstem sites averaged 0.131 ± 0.11 mg/L TKN as N in those three years. For the mainstem LCR sites, the percent difference between fall/winter low flows and spring/summer high flows in the three sampled years was 30% more TKN during high flows because they carry more detritus. The Kootenay site averaged 0.144 ± 0.11 mg/L TKN as N. Norn Creek averaged the lowest TKN of all sites, averaging 0.094 ± 0.09 mg/L TKN as N.

Total N includes nitrate, nitrite and TKN (organic N+ ammonia), and TKN was the largest Total N component in LCR. The mainstem sites averaged 0.161 ± 0.06 to 0.190 ± 0.17 mg/L T-N as N with the lowest total N at WQSI1 and the highest at WQSI3. The highest readings at each site were variable by season, with winters having the highest T-N. Total nitrogen donated in flows from ALR to LCR ranged from 0.147 to 0.216 mg/L in the upper 20 m (spill and release at HLK) during Apr-Nov 2014 (BC MoE data). This range is lower compared to LCR mainstem sites in 2014. Kootenay River had high T-N concentrations that averaged 0.19 ± 0.11 mg/L T-N, notably in winter, while Norns Creek T-N was low and averaged 0.098 ± 0.091 mg/L T-N as N.

We hypothesized that LCR water quality may be dependent on Arrow Lakes Reservoir (ALR) water quality because previous modelling of LCR temperature data showed a direct relationship with upstream conditions (Olson-Russello *et al.* 2014). Effects of nutrient enrichment at the closest ALR station (AR-8) upon LCR nutrients were investigated. Average monthly Dissolved Inorganic Nitrogen (DIN) at WQIS1 was positively correlated with average monthly DIN nutrient measurements at AR-8, ($r=0.56$, $p=0.005$ (data not shown)). Thus, DIN and T-N imported from ALR increase nitrogen concentrations in LCR. T-N results also indicate that there are additional nitrogen sources within the LCR above its confluence with Kootenay River that augment T-N concentrations in the flows from ALR.

1.5.5 Phosphorus

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

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

Inorganic ortho-phosphate (or SRP) represents the fraction of T-P that is readily available to periphyton for growth. In 2011 to 2015 samples from LCR, SRP never exceeded the detection limit of 0.01 mg/L, except at WQIS4, which is downstream of the Kootenay confluence and several municipal outfalls. Similarly, in flows from ALR, ortho-P/SRP seldom exceeded the detection limit.

The range of total phosphorus in ALR was <0.002 – 0.0034 mg/L in the upper 20 m during Apr – Nov 2014 (BC MoE data). The 2014 T-P averages were 0.003 ± 0.001 mg/L for ALR and 0.006 ± 0.004 mg/L for the mainstem LCR sites above the Kootenay confluence, indicating greater total phosphorus concentrations in the LCR. The correlation between average monthly T-P at WQIS1 and T-P nutrient additions at AR-8 was not significant ($r=-0.16$, $p=0.47$). Phosphorus sources within LCR may be as or more important than ALR nutrient additions. Mainstem total-P values ranged from <0.002 to 0.018 mg/L as P from 2008 - 2015. Winter and fall samples had the highest T-P, averaging ~0.006 mg/L, while spring and summer high flows offered more dilution and had about 30% less T-P. The concentrations of total phosphorus in the winter in LCR were variable between WQIS1 and WQIS3 (Figure 3-11). Operations such as Celgar and/or sewage outflows near these locations may affect the range in T-P values observed during winter low flows.

During this study, Norns Creek averaged ~0.008 mg/L T-P as P in winter, spring and fall, but only 0.003 ± 0.002 mg/L T-P as P in summer clear low flows.

The Kootenay River site averaged 0.007 ± 0.002 mg/L T-P as P during this study, slightly higher than LCR mainstem. The only season with a significant difference between the two rivers was Spring, where Kootenay samples had approximately double the T-P of LCR samples (Figure A19). Total phosphorus concentrations downstream of the Kootenay confluence reflected its concentration in a given season.

Winter T-P concentrations in 2013 (similar to unregulated flows) were not significantly different from 2014 and 2015 results (typical regulated flows), suggesting that T-P was not affected by variable MWF flows. In contrast, the rising leg of high freshet years may elevate spring particulate phosphorus but dilute dissolved phosphorus inputs, resulting in significant differences between RBT managed flow years (Figure A20).

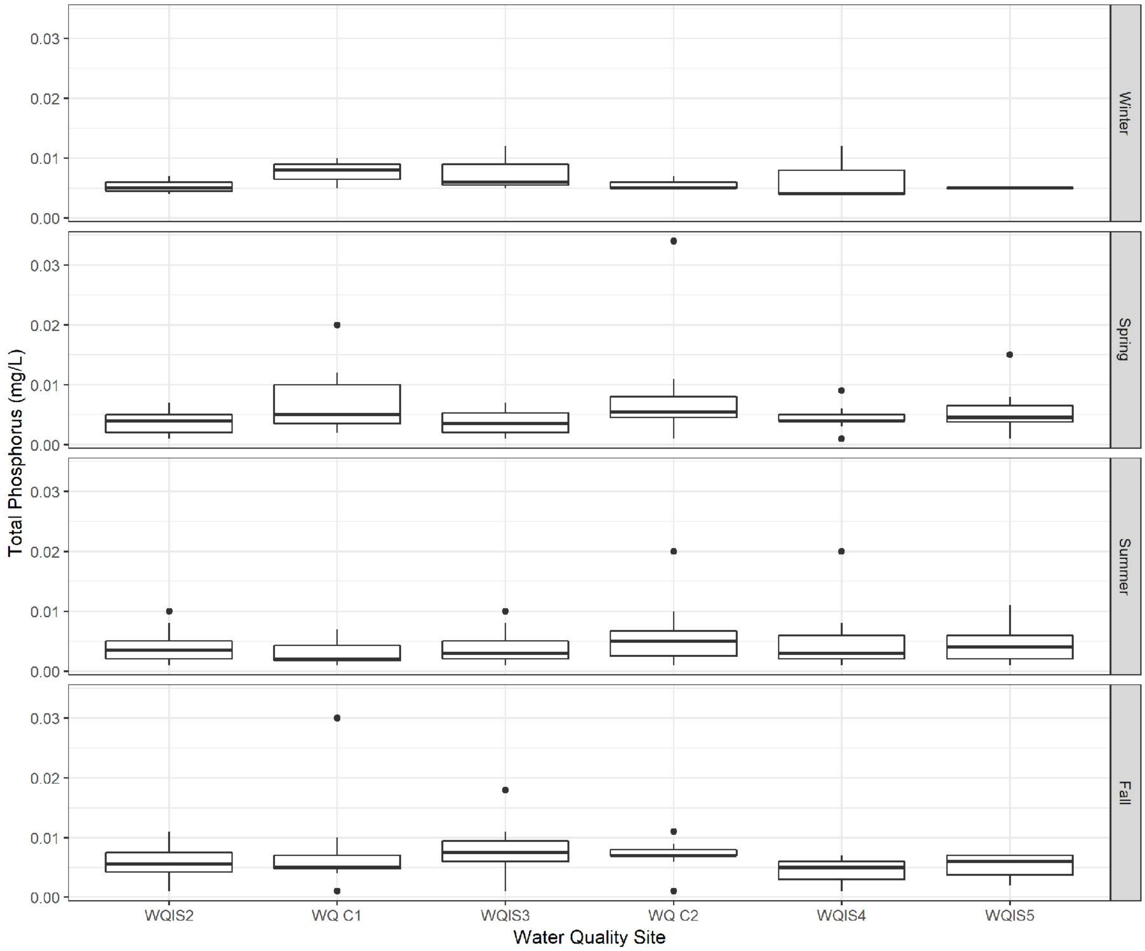

Figure A19. Total Phosphorus from LCR Water Quality Index Sites and Main Tributaries (2008-2015). BC MOE guideline is 0.005 - 0.015 mg/L for lakes; tentative LCR Objective = 0.03 mg/L T-P to avoid excessive algae growth.

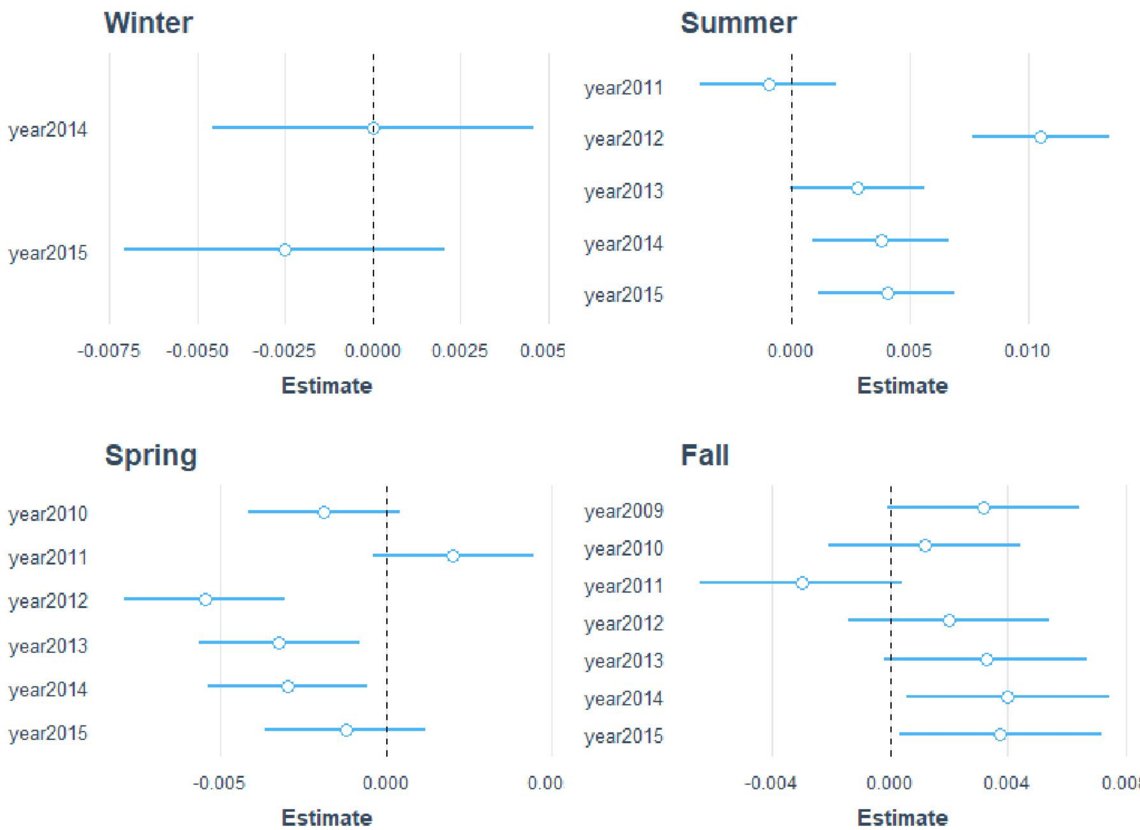

Figure A20. Fixed effects coefficients with 95% confidence intervals for year comparing Total Phosphorus from LCR Water Quality Index Sites. The reference year for winter=2013, spring=2009, summer=2008 and fall=2008.

The ratio of biologically available N and P is an important factor controlling river productivity. Globally, increased nitrogen loading is driving the world's largest rivers towards a higher DIN:DIP ratio through watershed disturbance, urbanization and the use of fossil fuels (Turner et al. 2003).

10.5.5 Turbidity

Turbidity measures how much sediment, organic detritus and organisms suspended in the water decreases its clarity. The range of turbidity measured in Kootenay and LCR flows was low and narrow. Turbidity measured in this study met BC guidelines protective of aquatic life. A turbidity spike would have to exceed background by 2 NTU for 30 days during clear flows or exceed background by 5 NTU at any time when background is 8 – 50 NTU during high flows to exceed the guideline (BC MoE 2012).

In LCR, turbidity collected in all flow periods in 2008 - 2015 except freshet and severe storms was within the range of previous years of 0.10 to 1.0 NTU (Figure A22). Turbidity spikes were observed during freshet flows of >7 NTU but they rarely exceeded 10 NTU magnitude (Figure A22). The turbidity range measured in ALR during Apr-Nov 2014 was 0.14 NTU in deep water to 0.38 NTU in shallow water (BC MoE data). This range is lower than the first station on LCR (0.1 – 1.2 NTU) indicating that there are turbidity sources within LCR (Figure A22).

As expected, the turbidity at Norns Creek was consistently higher in the spring compared to Kootenay River and LCR. Because the rivers are fed from reservoirs that allow settling of suspended materials, it logical that the turbidity values would be lower than unregulated Norns Creek flows. Seasonally averaged Kootenay flows ranged from 0.37 ± 0.35 NTU in fall to 3.6 ± 7.9 NTU in spring, while Norns Creek ranged from 0.46 ± 0.74 NTU in summer to 1.33 ± 1.37 NTU in spring 2008 – 2015 results. The highest turbidity measured in Norns Creek was still moderate at 4.1 NTU in freshet and 3.5 NTU immediately following a fall storm.

Winter and spring turbidity in LCR were stable and not significantly affected by flow management (Figure A21). Summer and fall turbidity were unusually high in 2015 due to sustained higher flows.

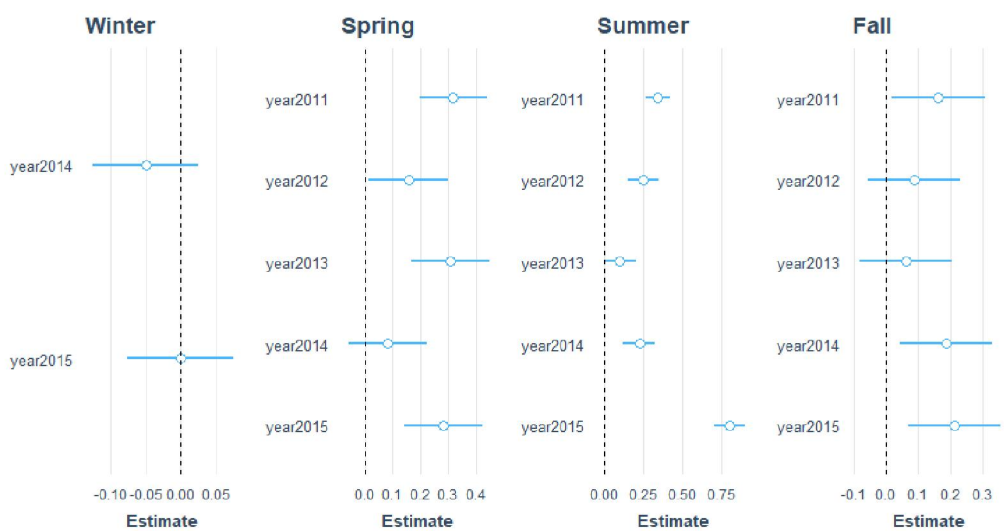

Figure A21. Fixed effects coefficients with 95% confidence intervals for year comparing turbidity from LCR Water Quality Index Sites. The reference year for winter=2013, spring=2009, summer=2008 and fall=2008.

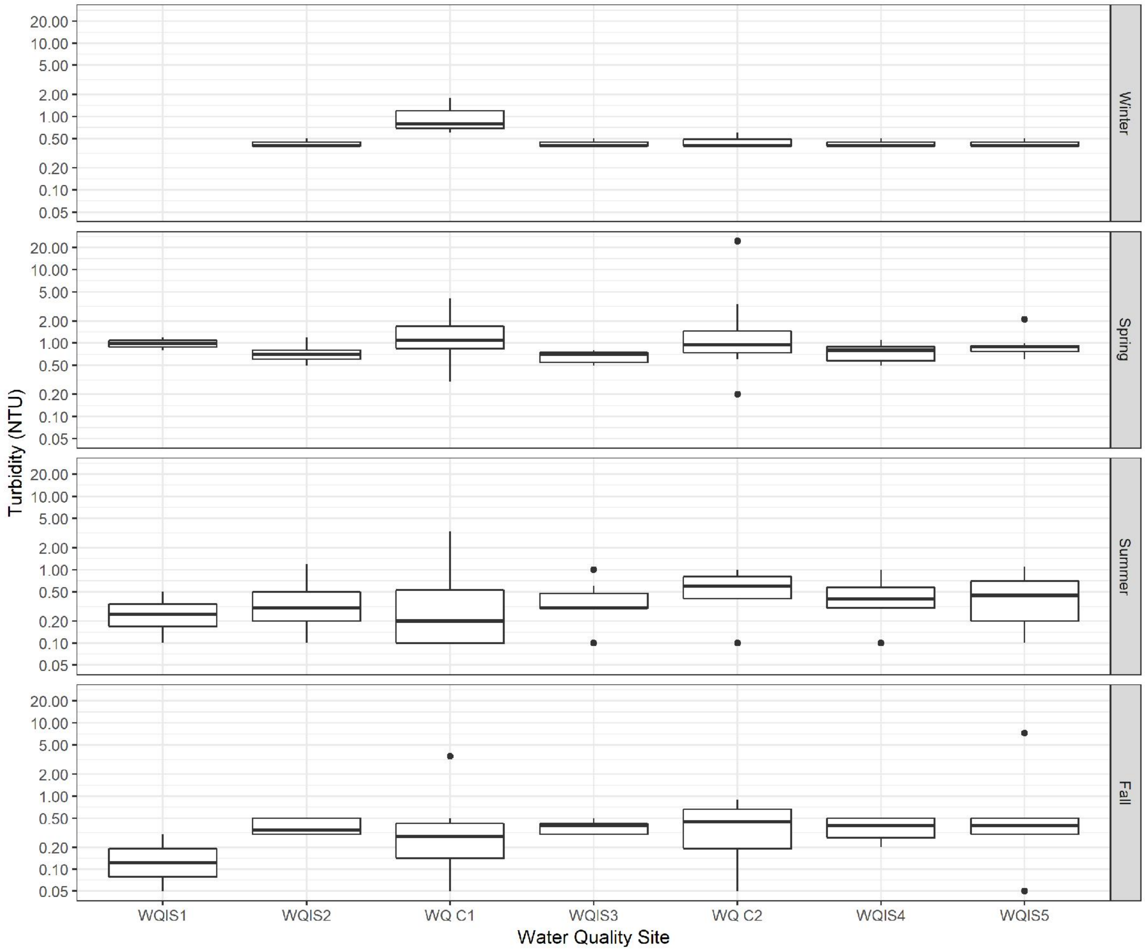

Figure A22. Turbidity from LCR Water Quality Index Sites and Main Tributaries (2008-2015). Aquatic life protection guidelines state maximum 24 hr increase = 8 NTU; maximum clear flow average (30 days) increase = 2 NTU.

10.5.6 Total Suspended Solids

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

Total suspended solids concentrations are typically low in the regulated LCR and Kootenay systems (Figure A23). Since all mainstem LCR samples consistently had TSS of less than 5 mg/L, a guideline exceedance involving a TSS spike of 25 mg/L for a duration of 24 h in clear flows, or an increase of 5 mg/L for a duration of 30 days in clear flows, could only occur in extreme flood. Like turbidity, the BC guidelines protective of aquatic life for TSS are unlikely to be exceeded at LCR mainstem sites. Most LCR TSS samples were non-detectable (< 1 mg/L) with rare exceptions.

Flows associated with freshet and storm flows were likely the contributing factor to TSS variability in LCR and its tributaries, but higher values were not recorded at all sites in heavy freshet years. Overall TSS was higher in Kootenay River, and at the sites downstream of Kootenay (WQIS4 and WQSI5) than overall TSS in the mainstem LCR. The highest recorded mainstem TSS value was 7.3 mg/L recorded below the Kootenay confluence during increasing flows at WQIS5 (Figure A23).

Overall TSS was higher in Norns Creek than the larger LCR. The TSS concentration recorded during spring 2012 at this small tributary was 15 mg/L. Similarly, fall 2014 storm flows caused an elevated reading of 11.5 mg/L TSS.

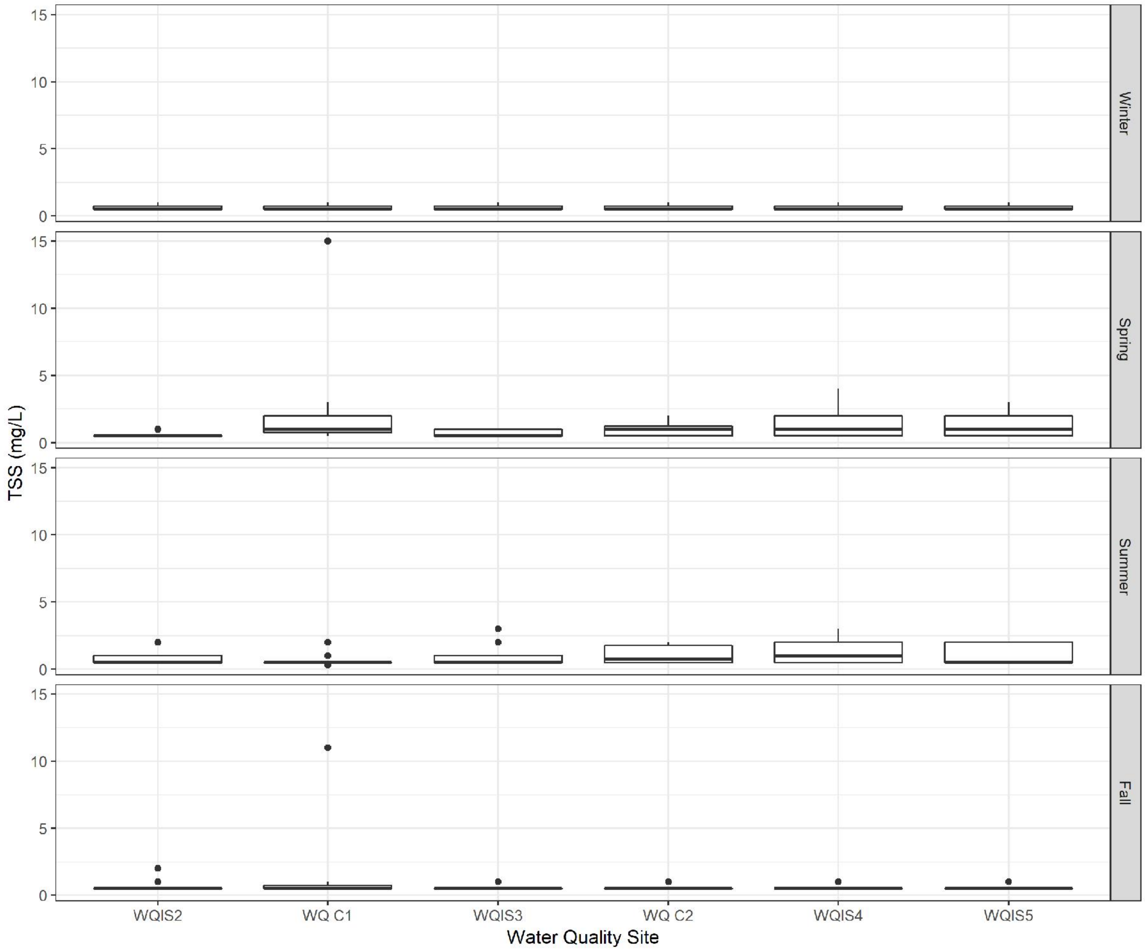

Figure A23. Total Suspended Solids from LCR Water Quality Index Sites and Main Tributaries (2008-2015). A number of measurements were estimated based on lab tolerance limits. No guideline or objective available.

10.5.7 Dissolved Oxygen

Dissolved oxygen enters water through turbulent flow, gas exchange and by photosynthesis. The capacity of water to hold dissolved oxygen is a function of its temperature. Dissolved oxygen concentrations were adequate for all salmonid life stages throughout this study (BC MoE 2012), and usually exceeded the 10 mg/L DO objective set for LCR (Butcher 1992), even in warm summer low flow conditions.

Dissolved oxygen in LCR mainstem sites ranged from 8.0 – 12.8 mg/L over the 2008 – 2015 growing seasons. Throughout the study period, dissolved oxygen declined during the summer in response to increased water temperature but did not fall below the 9.0 mg/L DO Objective at the mainstem sites (Figure A24). The LCR mainstem summer and fall readings had the lowest average DO range of 9.78 – 10.1 mg/L, while winter and spring had the highest at 10.7 – 11.6 mg/L. All summer dissolved oxygen samples met the 9.0 mg/L DO guideline at every mainstem site.

Dissolved oxygen in the Kootenay River consistently ranged from 7.0 mg/L in summer to 13.9 mg/L in spring with 118% saturation during spring high flows over the course of this study (Figure A24). Norns Creek is the second largest tributary to LCR and it measured from 7.1 mg/L in summer to 10.5 mg/L DO in winter (Larratt *et al.* 2013; Scofield *et al.* 2011). Readings were taken from within 1 m of the substrate in Norns Creek and averaged 100% oxygen saturation. Almost all summer dissolved oxygen samples met the 9.0 mg/L DO guideline in these two tributaries.

Mainstem LCR dissolved oxygen saturation ranged from a minimum of 81% in summer to a maximum of 122% in spring measurements. Percent saturations above 100% occur naturally with turbulence or when photosynthesis contributes oxygen that super-saturates the water. During this study, dissolved oxygen super-saturation has only been documented in the spring and summer months. The shift to sampling during the late fall and winter in 2012 resulted in a lower mean DO than what had been documented in previous years of the study when sampling was concentrated in the growing season.

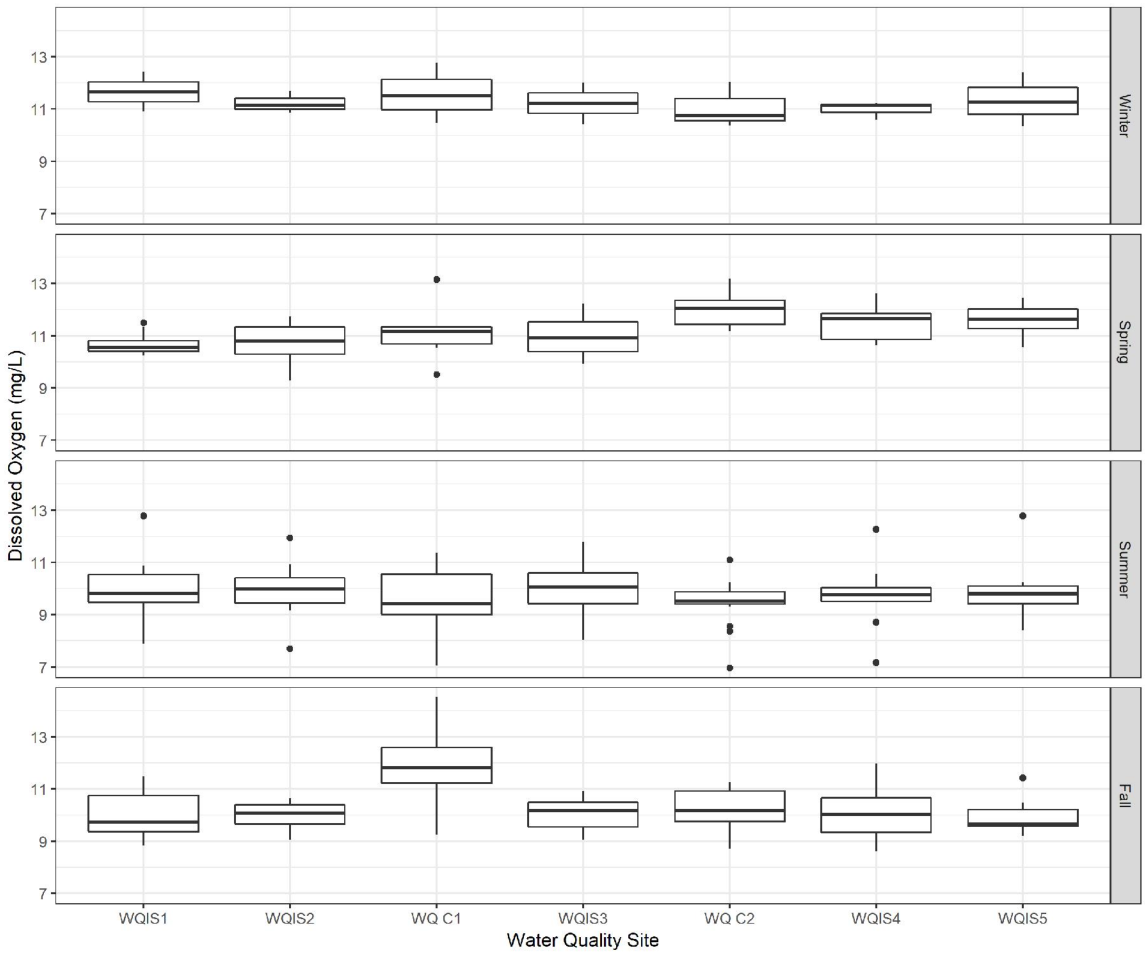

Figure A24. Dissolved Oxygen from LCR Water Quality Index Sites and Main Tributaries (2008-2015). BC MOE guideline is 9 mg/L; LCR Objective is 10 mg/L.

10.6 Discussion

LCR water quality sampling was intended to provide an understanding of river water chemistry and its influence on benthic productivity. Water quality sampling occurred monthly between June and October from 2008 to 2011, but was modified to coincide with biological sampling that occurred once per season from 2012 to 2015. Even with 8 years of data, it was difficult to test effects of flow on water chemistry as there were only 4-5 point samples taken each year. This makes addressing the water quality hypotheses (HO_{3phy} HO_{3Aphy} HO_{3Bphy}) challenging. The hypotheses for water quality state that *the continued implementation of MWF and RBT flows during winter and spring, and fluctuating flows during fall do not alter electrochemistry and biologically active nutrient concentrations in LCR*.

Overall LCR water chemistry is set by HLK and BRD dam releases which together account for 98% of flows. Most water quality parameters varied between years and exhibited distinct seasonal patterns due to flow, notably freshet. However, some water quality parameters did not appear to vary with flow. For example, both the Kootenay and Columbia systems showed stable pH throughout the study, even during the record 2012 freshet (Larratt *et al.* 2013). Fish flows should therefore have a minor influence on pH.

For those parameters that did vary with flow, the effects were frequently in proportion to the flow event. For example, the sediment carrying capacity of flowing water is proportional to its velocity (Hei *et al.* 2009; Giller and Malmqvist 1998; Hem 1985). For this reason, both turbidity and total suspended solids (TSS) were low during winter and fall flows on the regulated Columbia and Kootenay Rivers, with modest spikes during freshet and major storms. Although turbidity and TSS never exceeded guidelines and are unlikely to do so, they do affect light penetration, particularly into deep water. At the moderate turbidity levels found in LCR, light penetration to the shallow substrates would not hinder photosynthesis (Caux *et al.* 1997; ENSR 2001). However, the turbidity and light data collected during this study indicate that light penetration through water deeper than about 4 m would be reduced enough to influence periphyton production in LCR. Since the fish flows moderate flow variability in their respective hydrographs, peak flows that increase turbidity and suspended solids should be less frequent and may improve periphyton production.

Electrochemistry parameters in LCR tend to have an inverse relationship with flow, where higher freshet years had lower conductivity than in moderate freshet years (Olson-Russello *et al.* 2014). TDS tended to increase as water travelled through LCR and that increase was most evident in the fall and winter low flow periods. Spring flows reduced conductivity, principally through dilution and reduced groundwater inflows. In large freshet years such as 2011 and particularly 2012, conductivity was significantly lower. After freshet, lower late summer and fall flows and water elevations usually occurred, resulting in less dilution. During these lower flow periods, there is presumably a greater contribution from groundwater into the base flows which would increase electrochemistry (Peterson and Connelly 2001; Toulan *et al.* 2009; Golder 2010). Therefore, periods where flows are manipulated for fish should act on electrochemical parameters, particularly during significant departures from average unregulated flows. The winter 2013 MWF flows resemble the pre-management flows and provide an opportunity to compare winter flows in 2013 with 2014 and 2015. Of the studied parameters, conductivity in 2013 was significantly different than the other two MWF flow periods.

In any river system, there are numerous correlated influences on the biologically active (inorganic dissolved) nutrient concentrations. Dissolved nutrient concentrations in LCR are affected by factors including the limnology and nutrient status of Arrow Lakes Reservoir (Hatfield, 2008) and Kootenay Lake along with their respective fertilization programs, the numerous outfalls that exist on LCR, tributary and groundwater inputs, and regional land use, particularly septic fields and agriculture.

An effect of flow on total nutrients was observed and is likely the result of suspended organic detritus, sediment, phytoplankton and dislodged periphyton as measured in T-P, TKN and T-N. For example, total phosphorus concentrations spiked in 2014 immediately following a storm because more particulates were scoured into suspension. Similarly, organic nitrogen (TKN) increased during high flows by 25% in LCR and by 46% in Kootenay River. If the observed response of LCR to freshets and storm flows is broadly applicable, then moderately increased flows may improve the delivery of nutrients to periphyton. However, very high flows can increase scour and lower productivity while increasing total (biologically unavailable) nutrients. Large fluctuations in managed flows may increase total nutrients in the LCR, similar to the observed effect of a storm event. The influence of flows on total particulate nutrient concentrations must also consider imports from upstream Arrow Lakes and Kootenay reservoirs, particularly for total phosphorus, because algae cells exported seasonally to LCR will increase T-P. However, the influence of nutrient enhancement on LCR was not statistically detectable for T-P, indicating that there are additional nutrient sources affecting the LCR.

Unlike flow-induced scour affecting total nutrient concentrations, anthropogenic dissolved phosphorus sources are independent of flows and are therefore more likely to be diluted by higher flows. Throughout this study, ortho-phosphate (or SRP) rarely exceeded the detection limit of 0.01 mg/L in LCR samples, indicating oligotrophy. However, biologically important quantities of SRP are probably still present in the LCR as indicated by its stable, diverse periphyton populations. The SRP results are all lower than the historic range recorded for Birchbank and continue to follow a declining trend in LCR over the years (Holmes and Pommen 1999). During high freshet years, more inorganic nitrogen was observed in LCR than in years with lower peak flows (Scofield *et al.* 2011; Larratt *et al.* 2013). $NO_3 + NO_2$ and total phosphorus concentrations in LCR generally increased with greater flow variability. This may be caused by factors such as nutrient release from decomposition of organics in the varial zone, or variable groundwater influx. Within each year, seasonal effects occurred where summer nitrate concentrations were lowest and winter concentrations were highest. The Kootenay River is an important nitrate source in winter and spring. Amounts of nitrate that would be considered stimulatory to periphyton occurred more often during winter low flows when groundwater inflows would be important to base flows.

LCR temperature and dissolved inorganic N (DIN) data showed a direct relationship with upstream conditions in Arrow Lakes Reservoir (Olson-Russello *et al.* 2014), where DIN imported from ALR can elevate DIN in LCR. This has implications for managed flows because point source nutrients would be diluted during peak flows, while imported nutrients would not.

The results from this water quality study clearly identified flows as an important factor directing electrochemistry as well as both nutrient concentrations and nutrient delivery. Although no water quality samples for this study were collected prior to the implementation of managed fish flows, we can infer that the impact of flows on water chemistry in general will also apply to the difference in the hydrographs between the managed and unmanaged flows in the same period. The unmanaged flow hydrographs prior to the early 1990s were more variable than the managed hydrographs, with more extreme low flows that exposed shallow substrates and with higher peak flows that encouraged scour and diluted nutrient inputs.

Over the course of this study, one flow period resembled pre-managed fish flows. The winter 2013 MWF flows provide an opportunity to compare water chemistry during managed and unmanaged flows. Of the studied parameters, conductivity and nitrate (DIN) in 2013 were significantly elevated compared to the other two MWF flow periods, while other biologically important parameters such as T-P were not. Similarly, 2012 RBT freshet flows were unusually high compared to 2008–2011, 2013–2015 flows. Water chemistry parameters that differed during these extreme spring flows include significantly lower conductivity, higher nitrate and lower T-P concentrations compared to other RBT flow periods. These water chemistry shifts with flow events are the result of the balance between inflowing reservoir water quality and flow-directed dilution, scour and groundwater influx within the LCR.

Inferring what water chemistry might have been during unmanaged flows is not a substitute for data and prevents a formal pre/post implementation analysis. Additional constraints on statistical modelling emerged because sampling was concentrated in the growing season. Within each season, the water quality data was collected in a small range of flow conditions. The effect of managed flows could not be statistically test because there was limited variability of managed flows.

10.6.1 Impacts of Water Quality Changes on Periphyton

Based on data collected throughout the study, LCR has good water quality and limited biologically available nutrient concentrations indicative of oligotrophy. Parameters rarely exceeded water quality guidelines or objectives.

The biologically available nutrient data indicated that nitrate + nitrite and SRP concentrations were capable of influencing periphyton production in the LCR. Fish flows may improve particulate and dissolved nutrient delivery under stabilized, less variable flow conditions relative to unmanaged flows, but they are unlikely to alter the overall nutrient status of LCR. For example, winter low flows frequently had greater nutrient concentrations and greater periphyton productivity than high flow periods.

The data for turbidity/TSS indicated that modest turbidity spikes occurred during peak flows and they would shade deep water, altering the periphyton community.

Other parameters were less likely to exert a measurable influence on LCR productivity. For example, fish flows caused small decreases in electrochemistry parameters through dilution, and pH was stable throughout the flow periods. Similarly, dissolved oxygen increased during high flow periods but would not affect the periphyton community.

10.6.2 Summary of Water Quality Management Question/Hypotheses

Water quality sampling was undertaken from 2008 to 2015 to address Physical Habitat Monitoring Management Question 3. Data restrictions prevented before/after and regression modelling approaches to this management question. Instead, we compared the average hydrographs of unmanaged flows and fish flows to extrapolate our knowledge of flow impacts on LCR water chemistry. We also compared the data from seasons that resembled unmanaged flows to typical fish flows. Using this approach in this final report has led us to question our previous tentative acceptance of the management hypotheses $Ho_{3phy,}$ $Ho_{3Aphy,}$ and Ho_{3Bphy} (Olson-Russello et al. 2015).

We reject the management hypothesis $HO_{3Aphy,}$ that states that managed MWF flows have no effect on the water quality of LCR. The lines of evidence to support this rejection of hypothesis Ho_{3Aphy} include:

- the comparison of 2013 MWF flows (similar hydrograph to unmanaged flows) with 2014 and 2015 showed elevated conductivity and nitrate concentrations.
- Descriptive statistics suggested that MWF managed flow periods may influence total phosphorus concentrations
- Operations such as Celgar and sewage outflows can increase the range of T-P values observed during winter low flows (low dilution), and this would be evident in the first half of the MWF flow period.
- T-N results indicate that there are additional nitrogen sources in the LCR that augment concentrations in the flows from ALR above its confluence with Kootenay River. These sources would experience less dilution with managed MWF flows than unmanaged flows over the same period.

We reject the management hypothesis $HO_{3Bphy,}$ stating that managed RBT flows have no effect on the water quality of LCR. The lines of evidence to support this rejection of hypothesis Ho_{3Bphy} include:

- Descriptive statistics indicated that flow variability increased the availability of some nutrients (NO_2+NO_3 and total P), and RBT flows restrict flow variability.
- Descriptive statistics suggested that RBT managed flow periods may influence total phosphorus concentrations.
- The comparison of 2012 RBT (extreme freshet) to the other RBT flow years showed significantly lower conductivity, higher nitrate and lower T-P concentrations compared to other RBT flow periods.
- The stabilized RBT flows should lower organic and total nitrogen concentrations by reducing scour.
- Turbidity and TSS are positively correlated with flows, so lower peak velocities in RBT flow periods would limit turbid conditions.

We reject the management hypothesis Ho_{3Cphy}, that states that managed FFF flows have no effect on the water quality of LCR. The lines of evidence to support this rejection of hypothesis Ho_{3Cphy} include:

- Substrate exposure occurs during very low flows and can affect rates of groundwater influx and organic decay, particularly at shallow and mid shallow sites and substrate exposure was reduced under the first half of the fall fluctuating flow period.
- The higher averaged flows in the first half of the FFF period should decrease electrochemistry concentrations through dilution but increase organic and total nutrient concentrations through scour.

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11.0 APPENDIX 6. Ecological Productivity Monitoring - Management Question #1

11.1 Introduction

This appendix further addresses the ecological productivity monitoring management question #1 and relevant hypotheses.

MQ#1: *What is the composition, abundance, and biomass of epilithic algae in LCR? 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 epilithic algae?*

HO_{1eco}: *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_{1Aeco}:* *Continued implementation of MWF does not increase total biomass accrual of periphyton in LCR.*
- *HO_{1Beco}:* *Continued implementation of RBT flows does not increase total biomass accrual of periphyton in LCR.*
- *HO_{1Ceco}:* *Continued fluctuations of flow during the fall do not increase total biomass accrual of periphyton in LCR.*

11.2 Methods

11.2.1 Data Collection

Periphyton productivity was determined with the use of artificial substrates placed at seven sampling sites (S1-S7) within Reach 2 during three seasons (Figure A26). Periphyton sampling in later years of the study differed from Years 1-3, in that all sampling locations were in Reach 2. The reader should refer to annual reports for additional sampling locations and methodologies used during the first three years of the study (TG Eco-Logic 2009, 2010 and Scofield et al. 2011). In 2012, the transect depths sampled at each site were increased from three depths to five. Previously, depths were referred to as shallow [S], mid [M], or deep [D]. The five depths sampled since 2012 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 A15).

Table A15. Naming Convention of Sampling Depths and Corresponding Depth Strata

Depth Label	Depth Name	Depth Strata (m)	Years Sampled
D	Deep	>5.5	2008 – 2010, 2012, 2014, 2016, 2018
MD	Moderately deep	4 – 5.5	2012, 2014, 2016, 2018
M	Mid	2.5 – 4	2008 – 2010, 2012, 2014, 2016, 2018
MS	Moderately shallow	1 – 2.5	2012, 2014, 2016, 2018
S	Shallow	<1	2008 – 2010, 2012, 2014, 2016, 2018

In 2018, a single artificial sampler apparatus design was used for all seasons over a 10-week sampling duration (Figure 3-1). Winter samplers were deployed from January 9th through March 20th to coincide with the MWF flow period. The summer sampling period occurred from June 1st through August 9th and the fall sampling period occurred from August 9th through October 17th. The winter and fall sampling sessions entirely overlap with MWF and FFF flows, while only the first month of the summer deployment overlaps with the RBT flow period.

To ensure the samplers were deployed right side up, a chandelier method of deployment was used (Figure A25). Two ropes were fastened to the corners of the steel frame so that the periphyton sampler drifted through the water column horizontally. After the sampler was positioned on the bottom, the longest rope was pulled through the apparatus and back into the boat. Table A16 provides deployment dates, sampling numbers and equipment/sample recovery rates.

Figure A25. The Chandelier Deployment Method.

Figure A26. Reach 2 Benthic Productivity Sampling Locations.

Table A16. Periphyton artificial sampler deployment and recovery rates in 2018.

Season	Reach	Site	Periphyton Samplers	
			# Deployed	# Retrieved (% Recovery)
Winter (Jan 9 -Mar20) 10 weeks	2	Site 1 (S1)	5	5 (100)
		Site 2 (S2)	5	5 (100)
		Site 3 (S3)	5	5 (100)
		Site 4 (S4)	5	5 (100)
		Site 5 (S5)	5	5 (100)
		Site 6 (S6)	5	5 (100)
		Site 7 (S7)	5	5 (100)
Winter Totals			**35**	**35 (100)**
Summer (Jun 1 - Aug 9) 10 weeks	2	Site 1 (S1)	5	5 (100)
		Site 2 (S2)	5	5 (100)
		Site 3 (S3)	5	4 (80)
		Site 4 (S4)	5	5 (100)
		Site 5 (S5)	5	4 (80)
		Site 6 (S6)	5	5 (100)
		Site 7 (S7)	5	5 (100)
Summer Totals			**35**	**33 (94)**
Fall (Aug 9 - Oct 7) 10 weeks	2	Site 1 (S1)	5	5 (100)
		Site 2 (S2)	5	5 (100)
		Site 3 (S3)	5	3 (60)
		Site 4 (S4)	5	4 (80)
		Site 5 (S5)	5	5 (100)
		Site 6 (S6)	5	5 (100)
		Site 7 (S7)	5	5 (100)
Fall Totals			**35**	**32 (91)**
2018 Totals			**105**	**100 (95)**

NOTE: Lower sampler recovery rates were mostly due to shallow samplers being pulled to shore by bystanders.

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

11.2.2 Winter Accrual Data Collection

2018 was the third and final year that winter accrual sampling was undertaken. It is designed to investigate periphyton biomass accrual rates and test management hypothesis Ho_{2eco}. However, unlike 2014, only MS, M, and MD were included in 2016 and 2018 accrual sampling. Each deployed sampler was retrieved from the river at 2, 4, 6, 8 and 10 weeks after deployment. A single periphyton punch was randomly collected from the Styrofoam and was immediately packed on ice and placed in the dark until they could be delivered to Caro Labs Kelowna for chl-a analysis. The samplers were then carefully returned to the river bottom.

11.2.3 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. Fresh, chilled punches were examined within 48-hours for protozoa and other microflora that cannot be reliably identified from preserved samples. H. Larratt tested Lugol's solution compared to freezing the Styrofoam and determined that freezing provided enhanced long-term viability. One of the two punches was therefore frozen and stored until taxonomic identification and biovolume measurements could be undertaken. Species cell density and total biovolume were recorded for each sample. A photographic archive was compiled for all LCR samples. Detailed protocols on periphyton laboratory processing are available from Larratt Aquatic.

Periphyton datasets from 2018 and previous years of the study (2008 – 2010, 2012, 2014, 2016) 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

11.3 Datasets

Table A17. Datasets used in the analysis of ecological productivity management question #1.

Name/Description	Data Source	Years Obtained
Periphyton	Styrofoam punches collected at each productivity sampler during each deployment session. Data included taxonomic identification, biovolume, abundance and chlorophyll-a	2008 – 2010, 2012, 2013(winter only), 2014, 2016, 2018
Chlorophyll-a Time Series	Data collected at a select number of productivity samplers throughout the deployment periods, either weekly or biweekly	2008 – 2010 (summer and fall), 2014, 2016, 2018 (winter only)
Light / Temp	Data collected at each productivity sampler during each deployment session	2008 – 2010, 2012, 2013(winter only), 2014, 2016, 2018
Velocity	Data collected at each productivity sampler twice per deployment period	2009 – 2010, 2012, 2013(winter only), 2014, 2016, 2018

11.4 Analysis

Non-metric multidimensional scaling (NMDS) was used to explore variation in periphyton community composition at the family level. The Bray-Curtis dissimilarity index was used, as it is sensitive to the variation of species that have smaller abundances (Clarke and Warwick 1998). To visually explore differences in community compositions, the NMDS scores for samples collected between 2008 and 2018 were plotted using R package ggplot2 (Wickham 2009). A permutational multivariate analysis of variance (PERMANOVA) was used to determine if there were significant differences in community compositions according to depth, site, season and year. The amount of variability in community composition was determined by calculating the partial R^2 from a permutational MANOVA. Both NMDS and permutational MANOVAs do not make assumptions of the variable distributions and relationships (Anderson 2001; Clarke *et al.* 2006). The NMDS analysis and permutational MANOVA used R package vegan version 2.3-5 (Oksanen *et al.* 2018). The NMDS analysis was performed with rare taxa included and excluded and both results were very similar. Rare taxa were defined as taxa that represented less than 5% of the total samples. The results presented are with rare taxa excluded. To identify taxonomic differences between samples, taxa were related to the community differences by fitting them to the ordination plot as factors using Envfit (Oksanen et al. 2018).

To better understand the periphyton productivity and community composition, the metrics in Table A18 were calculated. In addition, the percent of major periphyton groupings were also calculated. Trait-based periphyton ecological guild analyses were undertaken this year to help detect differences in community composition that were related to flow. The method developed by Passy (2007), and the planktonic guild (PG) developed by Rimet & Bouchez (2011, 2012) were employed. We applied the diatom guild descriptions to non-diatom algae. Based on the literature, we expected Low Profile guild to dominate areas of turbulence, scour, and hydrologic disturbance; High Profile guild to do best in regions with stable flows; and finally, the Planktonic guild was expected to do best immediately downstream of reservoir discharges. Use of the guild approach highlights large-scale changes and drivers as opposed to the more nuanced and complex approach of considering each taxa's distribution individually.

Table A18. **Responses for Periphyton.**

Variable	**Description**
Total Abundance	Total Abundance across all species
Total Biovolume	Total Biovolume across all species
Effective Number of Species	A measure of community diversity that is the e^S. S= Shannon-Wiener index.
Species Richness	Number of unique species
Percent High Profile Guild	The percentage of high-profile guild was calculated based on abundance and biovolume
Percent Low Profile Guild	The percentage of low-profile guild was calculated based on abundance and biovolume
Percent Planktonic Guild	The percentage of low-profile guild was calculated based on abundance and biovolume

Linear mixed effects models were used for the MWF flow period to compare annual variations in periphyton productivity and community composition. Data from the shallow (S) samplers was excluded from the analysis, because in some years they were prone to exposure due to low flows or tampering. Total biovolume and chl-a were log10 transformed to reduce heteroscedasticity in model residuals, whereas effective number of species did not require a transformation. The periphyton sampling design does not include true replicates. However, there is pseudo-replication among periphyton samples. The level of pseudo-replication is difficult to determine but it is expected that pseudo-replication occurs at the site level. In some cases, pseudo-replication may also occur at the site and year level within a given season.

The correct selection of a random effect is required to ensure the linear mixed effects model does not violate the assumption of non-independent observations. Separate models for the MWF flow period were fit with site as a random effect and site and year combination as a random effect. If the model that included site and year as a random effect had a lower AIC than the model that included site as a random effect it was selected for the final mixed effects model. For the total biovolume model site was used as the random effect, whereas the effective number of species model used the combination of site and year for the random effect. The combination of site and transect was used as the random effect for the chl-a model. The 95% confidence intervals for the fixed coefficient of year was calculated and plotted using the R package jtools version 2.0.1 (Long 2019).

Linear mixed effects models were also used to determine if the percent High Profile Guild, calculated from abundance, had annual differences within each of the managed flow periods. The percent High Profile Guild is a metric that is sensitive to flows. The periphyton data from 2008-2011 was not included in the comparison of the percent High Profile guild between years because only three transects were sampled in those years. Data from the shallow (S) samplers was also excluded from the analysis, because in some years they were prone to exposure due to low flows or tampering. The summer and fall linear mixed effects models used site as the random effect, whereas winter used the combination of site and year as the random effect. The 95% confidence intervals for the fixed coefficient of year was calculated and plotted using the R package jtools version 2.0.1 (Long 2019).

The total hours over 10 photons/m^2/sec was calculated from the light tidbit data for each periphyton sampler to confirm that all samplers were in the photic zone. The 10 photons/m^2/sec light threshold was chosen because it is based on the known light tolerances of periphytic algae (Sigee 2005). Periphyton productivity metrics are expected to increase with the total hours over 10 photons/m^2/sec. This is roughly 2% of full sunlight striking the water surface (<1% is usually accepted as the photosynthetic limit) (Jassby and Platt 1976; Hill and Fanta 2008).

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

11.5 Results

Periphyton sampling was focused on the most productive area of the river - the permanently wetted, shallow substrates in LCR Reach 2, from the water's edge to depths of 5 - 6 m. The samplers were distributed as widely as possible at each site but none could be deployed in the deepest thalweg areas that frequently exceeded 10 m depth. All the samplers were deployed in the LCR photic zone and received more than 10 photons/m^2/sec (Figure A27). Overall, periphyton growth in this key production area would classify LCR as moderately productive.

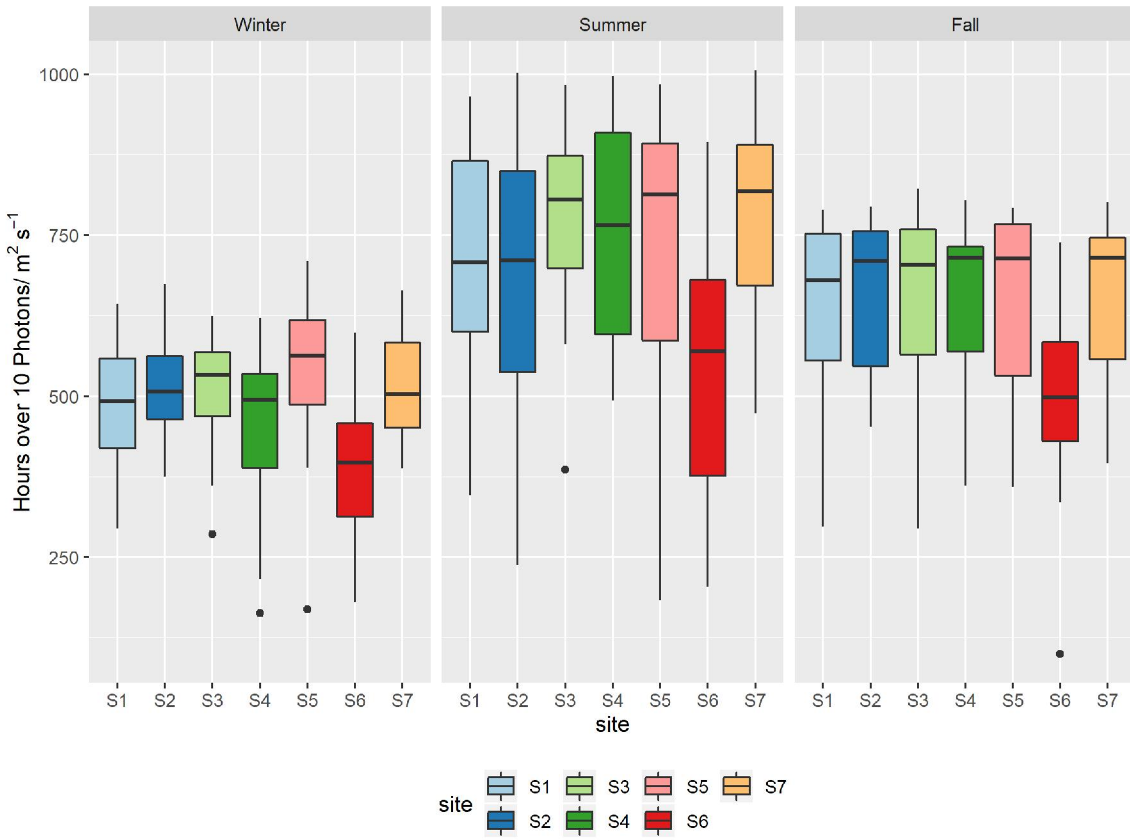

Figure A27. Total hours over 10 photons/m^2/sec for LCR periphyton samples for all years grouped by season and site.

11.5.1 Periphyton Community Structure

Diversity metrics (species richness, Simpsons index, Effective number of species) were similar over the transect span from shallow to deep (Figure A28). Average species richness ranged from 24 ± 6 to 46 ± 7 in all LCR samples, with an overall average of 32 ± 6 taxa. Species diversity and the Simpson's index results indicate that LCR periphyton biodiversity is stable. Periphyton diversity in LCR was far higher than the diversity observed in MCR despite a similar range of substrates. For comparison, MCR mean taxa richness was 18 ± 6 in the spring and 20 ± 6 in the fall (Larratt et al., 2017) (Digital Appendix B, 3-4).

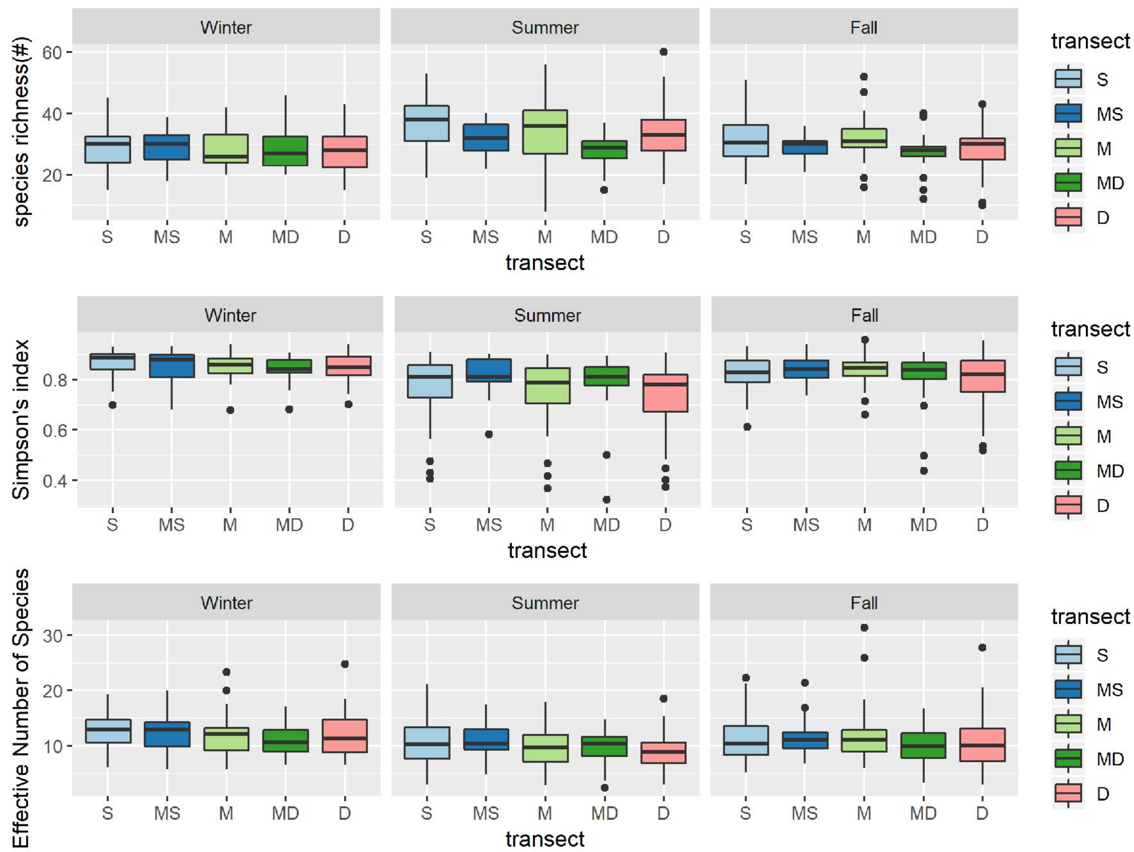

Figure A28. Community diversity metrics for winter, summer and fall in 2014, 2016 and 2018, over the range of sampled depths. Depth labels are: S=shallow, MS=moderately shallow, M=mid, MD=moderately deep, D=deep

A total of 76 periphyton algae taxa were frequently observed in LCR, 22 of which were common to all samples and 46 taxa occurred at less than 5% of the sites. Like most large rivers, LCR periphyton was dominated by diatoms representing between 55 and 99% of the average biovolume in all sample sites and seasons (data not shown).

Over the years of study, the largest shifts in community structure occurred in the soft-bodied algae. For example, flagellate abundance oscillated over the sample periods and ranged from 0 - 40%. Filamentous cyanobacteria ranged from 0.6 - 46% by abundance, but that translated to only 0.01 - 1.0 % of the total biovolume because of their small cell size. Large filamentous green algae are slower growing and occurred most often on the sides of stable cobbles where there is more protection from scour and shear. Seven taxa colonized or drifted onto the artificial substrates during the 8 - 12 week fall and summer deployments, accounting for 0 – 44% of biovolume. They were most common on the permanently wetted shallow substrates, and their abundance tended to decline with increasing water velocity. The nuisance diatom *Didymosphenia geminata* (Didymo) was detected at all LCR sample sites and was most prevalent in winter samples (21%) and lowest (5%) in the high-flow summer period.

Despite the moderate and stable production in LCR in years with typical flows, there were substantial differences in the periphyton community observed between the three seasonal deployments in LCR. The NMDS analysis showed significant differences in periphyton community composition between season and years (Table A19), particularly in the soft-bodied algae. For example, the flagellate Astasiaceae was positively associated with axis 1, whereas pico-flagellates were negatively associated with axis 1 (Table A20). This suggests that Astasiaceae are more abundant in winter than in summer and fall. The cyanobacteria *Synechococcacea* was negatively associated with axis 1 which suggests *Synechococcacea* is more abundant during the summer. Axis 1 represents the periphyton community differences in flagellates and cyanobacteria among years.

Table A19. PERMANOVA results for periphyton at Family level.

group	R_stat	Fstat	p_val
year	0.050	28.040	<0.001
season	0.130	39.140	<0.001
depth	0.020	2.370	0.002
site	0.020	1.650	0.005

Table A20. Taxa scores for NMDS axes with p-value and R^2.

Species	pval	r2	NMDS1	NMDS2
Achnanthaceae	0.000	0.230	-0.160	-0.450
Astasiaceae (flagellates)	0.000	0.250	0.500	0.030
pico-flagellates	0.000	0.320	-0.490	0.290
Stephanodiscaceae	0.000	0.450	-0.320	-0.590
Synechococcacea	0.000	0.320	-0.470	0.310

There were seasonal differences in the low and high profile ecological guilds because they are sensitive to flow. Taxa from the low profile guild (taxa that can withstand higher flows) contributed less to periphyton community in winter compared to summer and fall (Figure A30 and Figure A31). The percent of high profile guild abundance was higher in winter than in other seasons (Figure A30). However, the percent biovolume of high-profile guild was similar across seasons. In the winter, the furthest upstream sites, S1 and S2, had lower percent biovolume of high-profile guild because a large proportion of the biovolume was from planktonic diatoms from ALR.

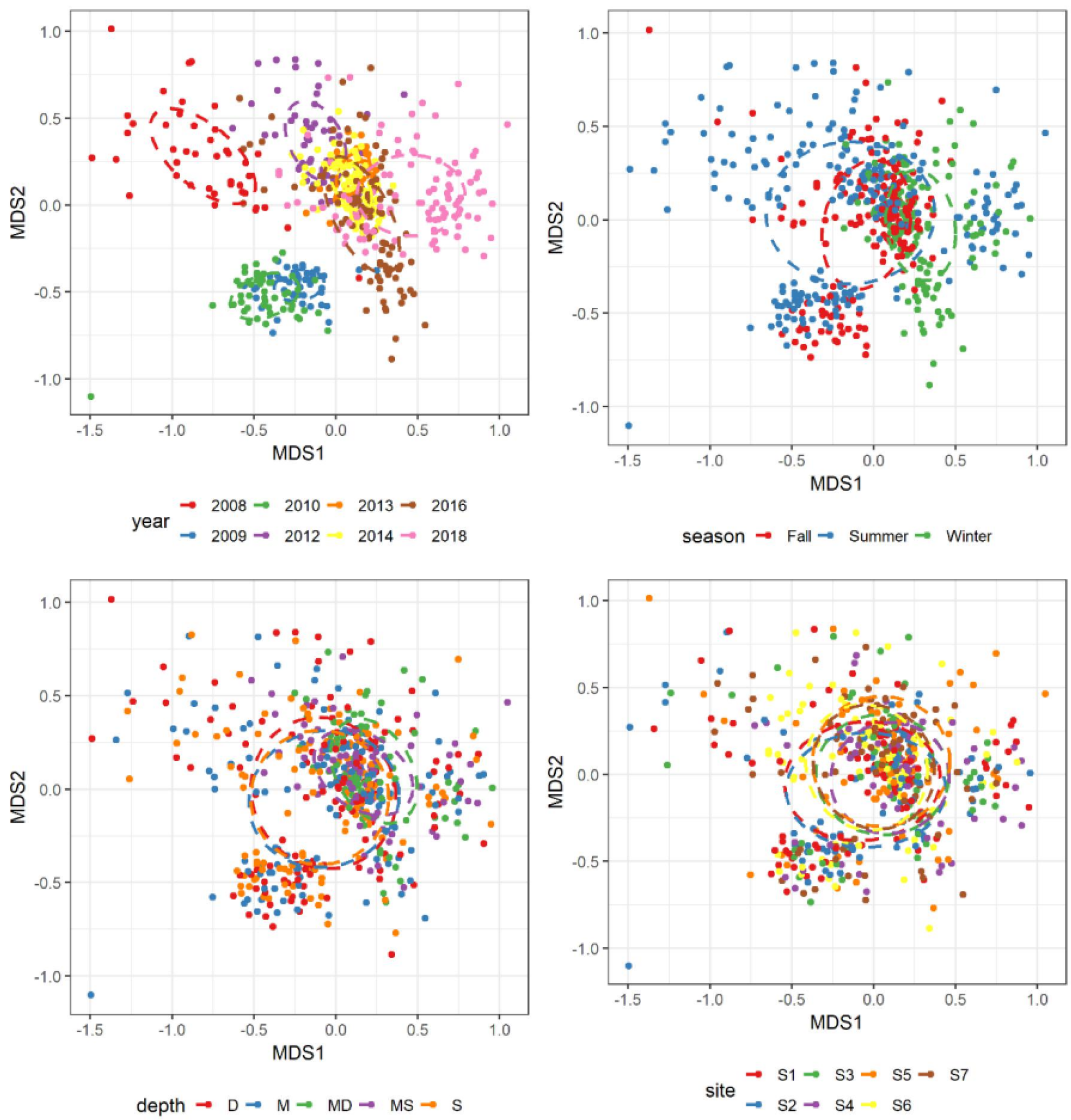

Figure A29. NMDS of periphyton family level abundance grouped by year, season, depth and site for all data from 2008 – 2018. The closer points are together the more similar the periphyton community composition is. The NMDS used a Bray-Curtis dissimilarity index and had a stress index of 0.23. Ellipses are calculated based on 95% confidence interval of the NMDS scores for each group.

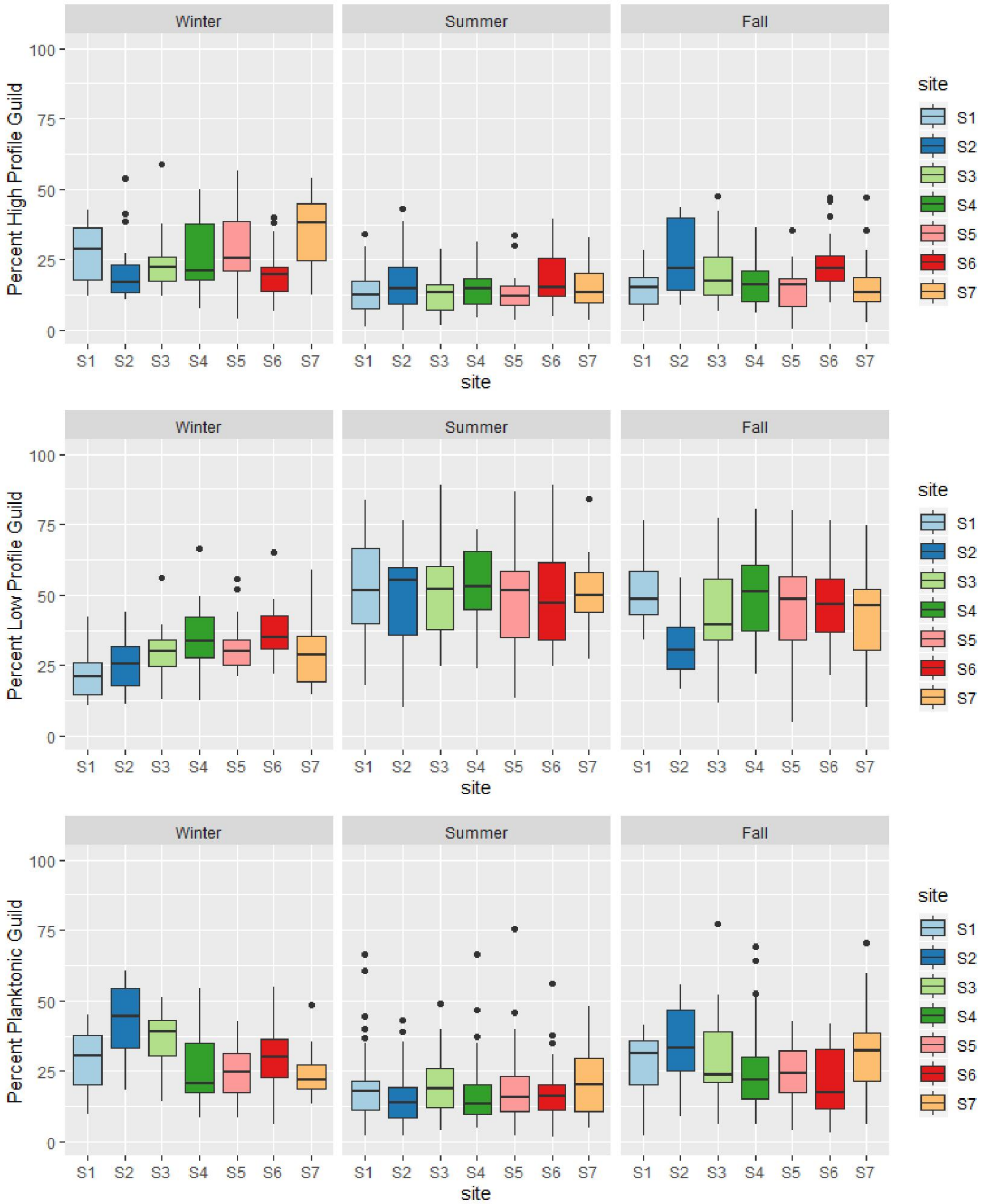

Figure A30. LCR periphyton community composition for all years using percent abundance guild by season

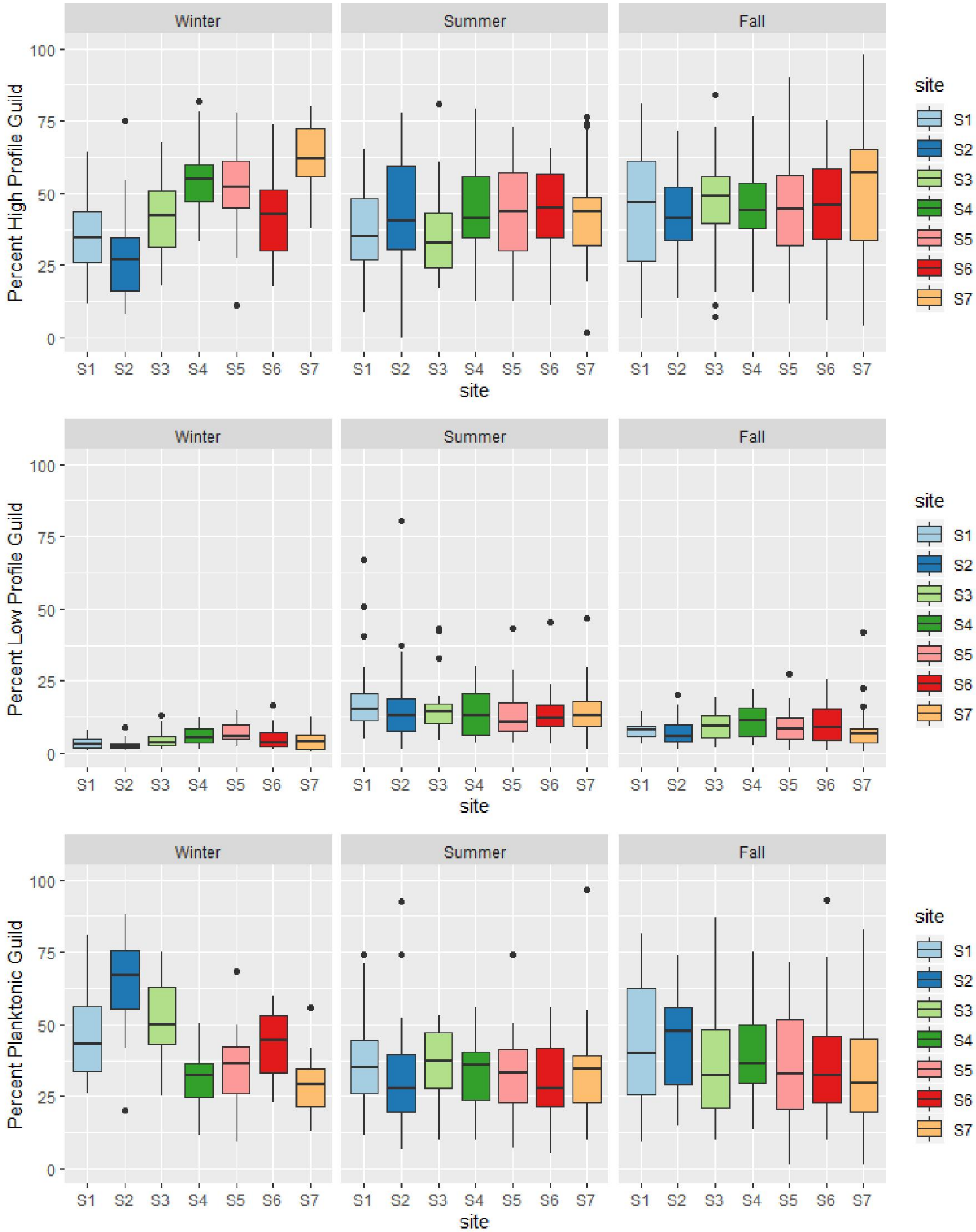

Figure A31. LCR periphyton community composition for all years using percent biovolume guild by season.

11.5.2 Periphyton Production – Standing Crop

Periphyton standing crop metrics of total biovolume, chl-a, and total abundance varied among seasons. The summer period which includes freshet flows had the lowest periphyton production across all years compared to other sample periods. The winter sampling period had higher periphyton biovolume compared to the summer and fall sampling periods (Figure A34). The biovolume results were affected by the higher occurrence of Didymo in winter. The winter and fall sampling periods had comparable chl-a and periphyton abundance. The highest values of chl-a in 2016 were seen in a few winter samples (chl-a >40 µg/cm^2). Other high values occurred in fall 2010 at S6, S7 and fall 2014 at S6 that had greater filamentous green algae growth under moderate flows. As expected, LCR production metrics for biovolume and chl-a were correlated (r=0.61, p<0.001).

Density of the nuisance algae Didymo (high profile guild) was highest in winter. Generally, sites with back-watering from the Kootenay River (S4, S5, and S7) had higher relative abundance of Didymo than the sites downstream of HLK dam, S1, S2, S3, and S6 (Figure A32).

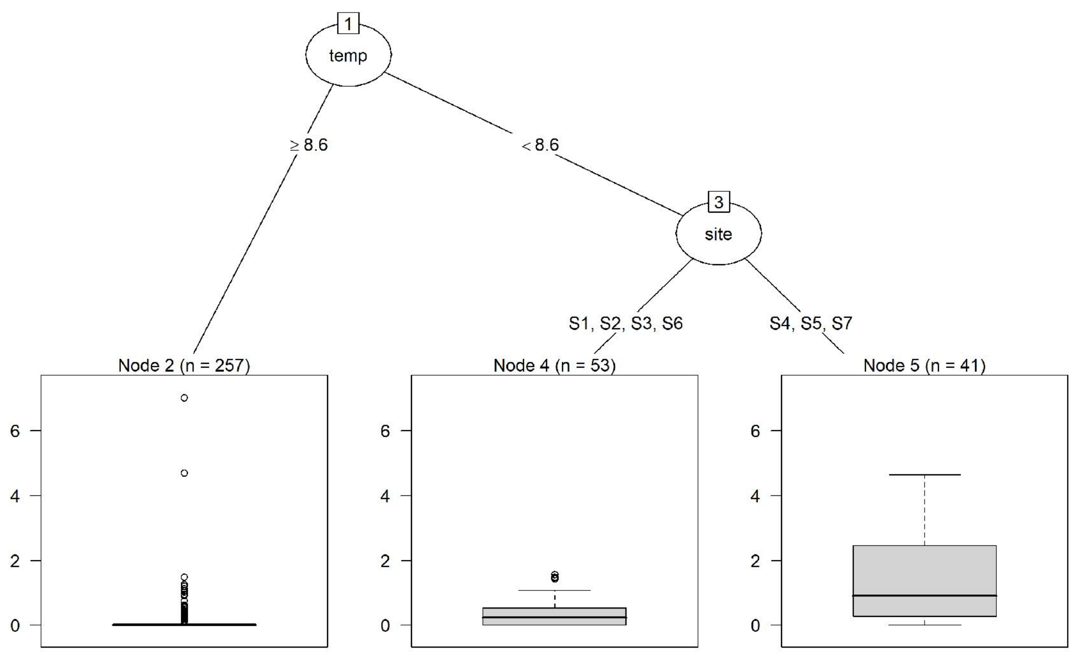

Figure A32. CART model for relative abundance of Didymo for fall, summer, and winter 2008-2018. The splitting rules indicate how the data is being grouped. The final groups (terminal nodes of the tree) show boxplots of percent relative abundance of Didymo. Temp is mean daily water temperature of the sampling period.

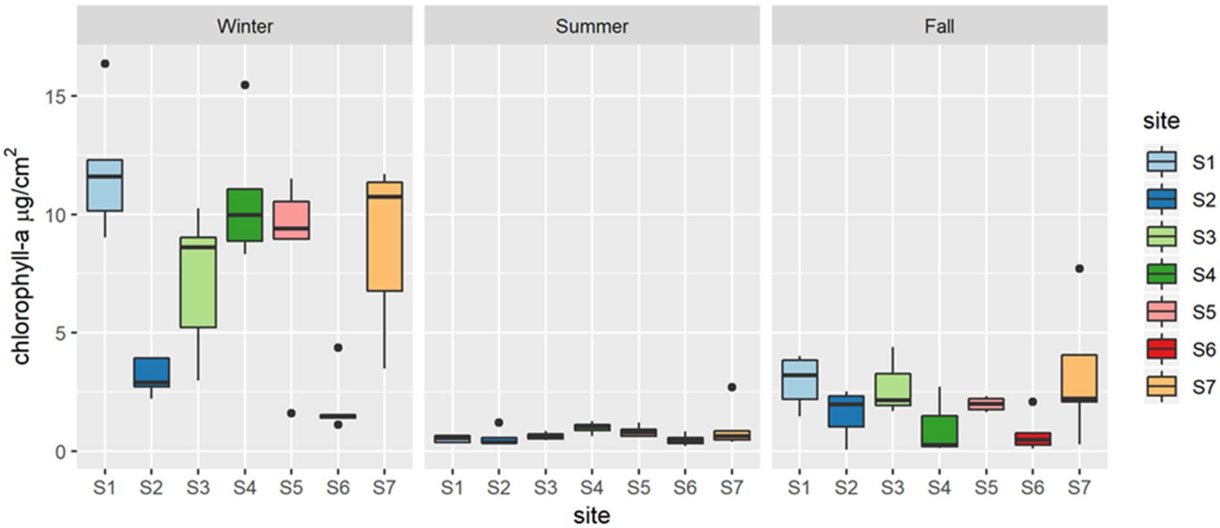

Figure A33. Chlorophyll-a (ug/cm^2) in winter, summer and fall in 2014, 2016 and 2018, across all sites.

Of the seven sample sites in R2, S6 consistently had the lowest periphyton productivity across all seasons and it was the only permanently depositional site (Figure A33). Sites S1 and S7 had the highest overall chl-a, and the remaining mainstem sites were intermediate.

The transect depth where peak biomass occurred varied with season and sample site. Winter and fall modest flows showed distinct productivity peaks while summer high flows showed low productivity throughout. Over the years of study, the transect position with the greatest periphyton abundance, biovolume and chl-a was MS to M in winter and MD in fall (Figure A34).

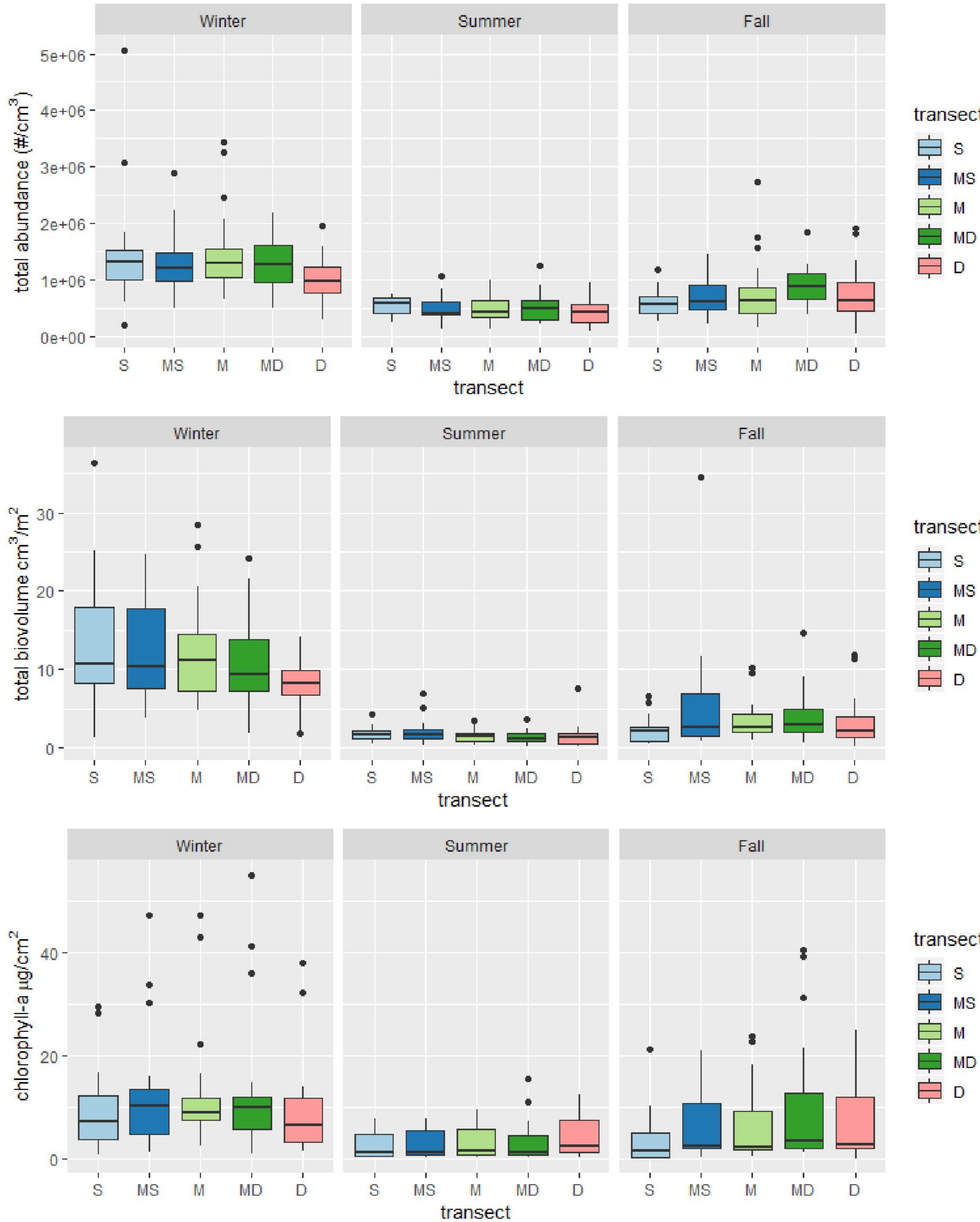

Figure A34. Periphyton abundance (cells/cm2) biovolume (cm^3/m^2) and chl-a (ug/cm^2) in winter, summer and fall in 2014, 2016 and 2018, over the range of sampled depths. Depth labels are: S=shallow, MS=moderately shallow, M=mid, MD=moderately deep, D=deep.

Abundance, biovolume and chl-a consider live periphyton while ash-free dry weight (AFDW or volatile solids) includes all live and dead organic material. Like other metrics, AFDW analyses confirmed that winter is by far the most productive period in LCR for periphyton, and again, Didymo growth was a key driver. Summer seasons with the freshet flow periods were consistently lowest for AFDW as they were for periphyton productivity metrics (Figure A35). Fall results for AFDW can be inflated by caddisfly biomass. Caddisfly biomass exceeded periphyton biomass in 2014 fall samples.

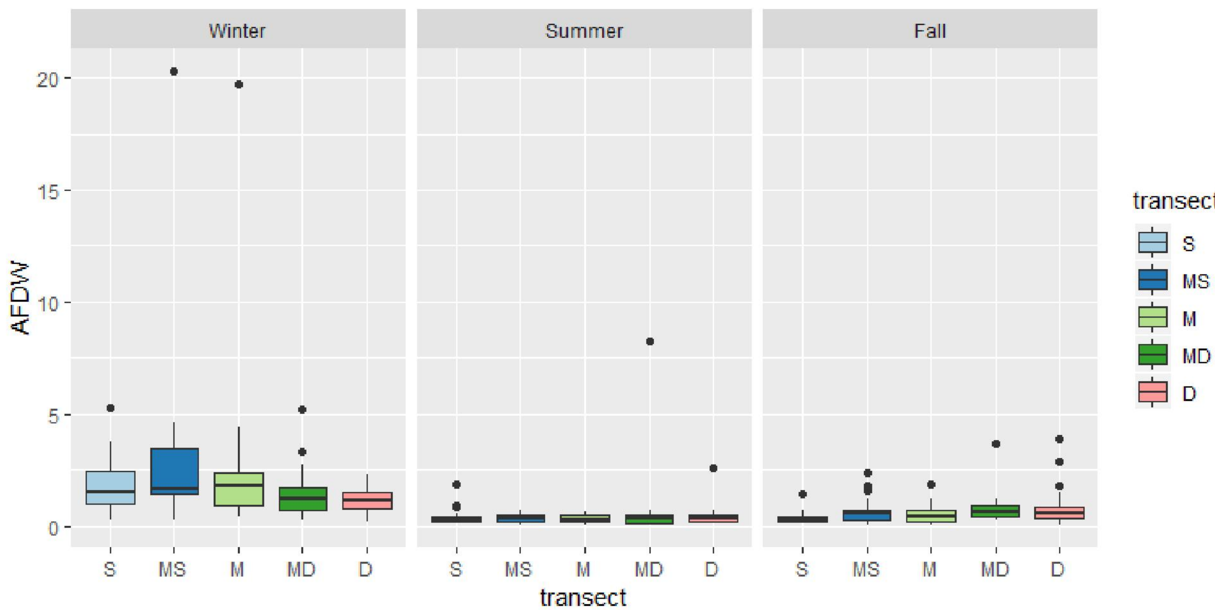

Figure A35. Ash-free dry weight (mg/cm^2) by season and year in 2014, 2016 and 2018 over the range of sampled depths**.** Depth labels are: S=shallow, MS=moderately shallow, M=mid, MD=moderately deep, D=deep.

11.5.3 Impact of Dewatered Substrates

Most of the substrate and sampler dewatering occurred during deployments in the FFF period. Over the studied years, sites with the most dewatering included S2, S3 and S4 at shallow and mid-shallow sites, with less frequent dewatering of S5, S6 and S7 shallow sites. Dewatering occurred during late September and October and lasted 2 to 76 hours (Table A21). Most shallow sites showed lower productivity and diversity during the FFF sampling period (Figure A34).

Table A21. Summary of exposure times and productivity metrics in Fall 2018

Sampler	Time Exposed (hours)	Abundance (cell/cm2)	Biovolume (cm3/m2)	Chl-a (ug/cm2)
R2 S2 S	62	489,552	3.48	0.07
R2 S3 S	171	-	-	-
R2 S4 MD	29	-	-	-
R2 S4 MS	198	248,724	2.20	0.25
R2 S4 S	62	27,1096	2.51	0.14
R2 S6 S	59	626,416	2.12	0.77
R2 S7 S	76	623,784	4.21	0.29

Shallow sites also showed the highest percent dead diatoms both in abundance and biovolume. For example, 2016 percent dead abundance decreased from 12% at shallow sites to 7.4% at deep sites (dead diatom biovolume decreased from 16.8% to 10.4%). Also, there was higher dead biovolume in years with frequent substrate dewatering during FFF (e.g., 2016, 2018), compared to years when there was less dewatering (e.g., 2014).

11.5.4 Periphyton Productivity Statistical Analyses

The three years of winter data together with the five years of full transect summer and fall data were used to statistically determine fish flow impacts on periphyton communities. No data were collected during the unmanaged flow periods, preventing a before/after analysis. Instead, reference years were selected that had similar flows to the unmanaged period. The reference year for MWF was 2013 where the hydrograph was similar to unmanaged flows, and for both RBT and FFF it was 2012, the record high freshet flow year (but otherwise, not similar to unmanaged flows).

Periphyton and community composition metrics for MWF were compared to 2013 using linear mixed effects models. The diversity of the periphyton community was significantly higher in 2013 compared to the other winters (Figure A37). The winter of 2016 was the only year to have significantly different chl-a than winter 2013, whereas winter 2014 was the only year biovolume was significantly different than winter 2013.

In MWF period, the percent high profile guild was significantly higher in 2014 and 2016 compared to 2013 (Figure A36). The lower flows in winter 2014 and 2016 provided favourable conditions for high profile taxa (Figure 4-2).

During the summer RBT flow period that includes freshet, shifts in periphyton community structure occurred that were likely induced by flows. For example, percent high profile guild was significantly higher in years with moderate RBT flows compared to the extreme flow year 2012 (Figure A36).

During the fall fluctuating flow period (FFF) the high profile guild may have benefitted from the relate FFF flow stability over unmanaged flows, but the data are too variable to draw conclusions from this line of evidence. (Figure A36).

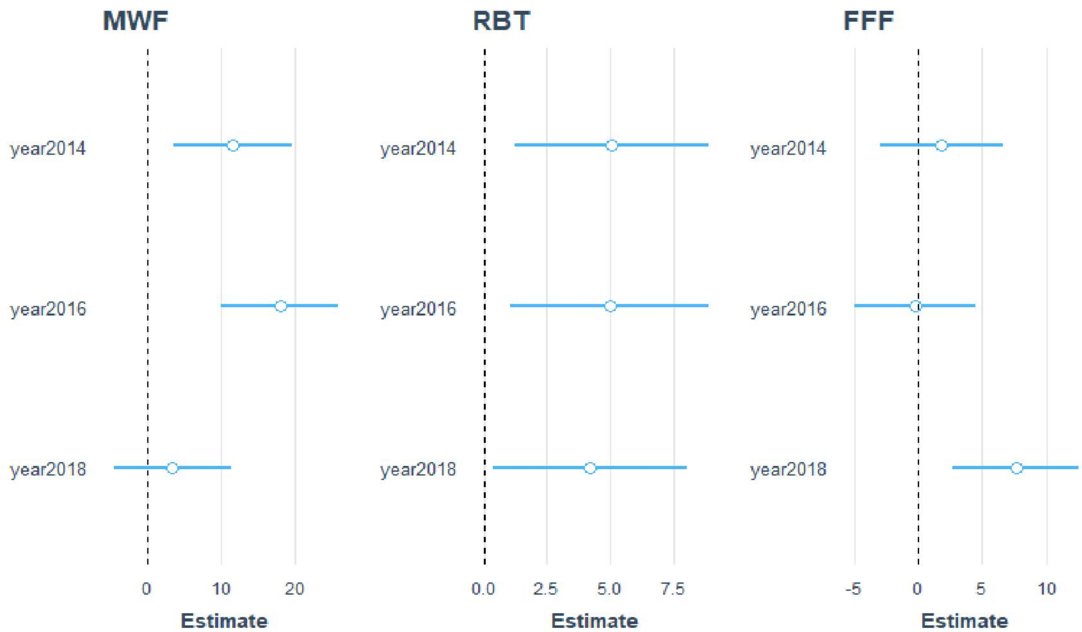

Figure A36. Percent High Profile Guild for MWF, RBT and FFF flow periods, fixed effects confidence intervals for year compared to reference year (MWF=2013 – similar to unmanaged flows; RBT and FFF=2012 – record high flows).

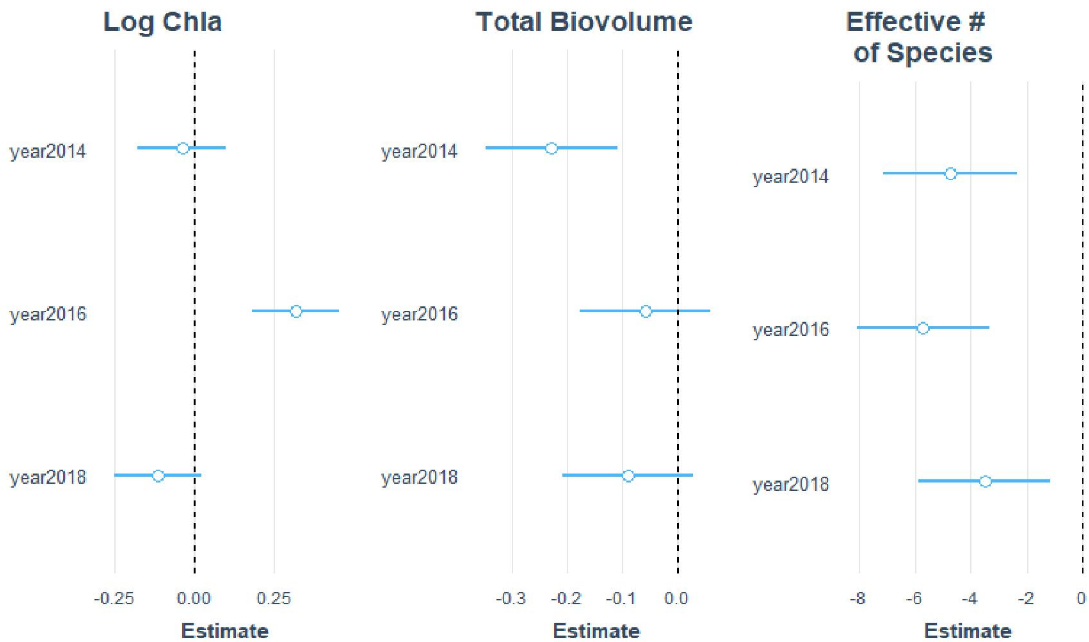

Figure A37. Periphyton Productivity and Composition for the MWF flow period, fixed effects confidence intervals for year compared to reference year 2013.

11.5.5 Periphyton Accrual

Although LCR periphyton standing crop metrics were variable between seasons and years, the chl-a time series data indicated that accrual reaches peak biomass in 6-7 weeks in summer, greater than 8 weeks in fall and greater than 10 weeks in winter (Figure A38). A few very long-term samples were collected in winter sessions. Mid-depth samplers were deployed in winter 2013 for 12 and 26 weeks and chl-a peaked at 12 weeks, although periphyton biovolume continued to climb. In the 2014 winter deployment, both chl-a and biovolume were lower at 20 weeks than at 10 weeks. When samplers were deployed for longer than these periods, the standing crops stabilized or declined.

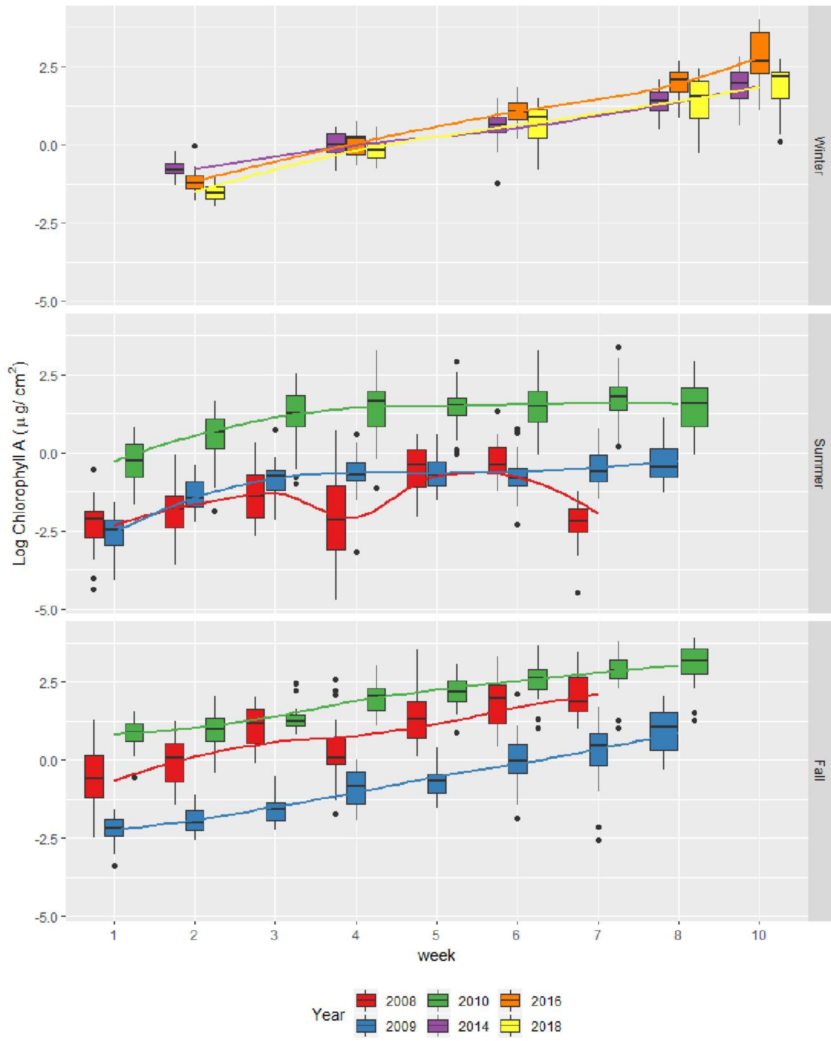

Figure A38. Weekly time series periphyton chl-a accrual rates in summer (2008 – 2010), fall (2008 – 2010) and winter (2014, 2016, 2018). Fitted lines were generated using a locally weighted polynomial regression method (LOWESS). The first three years of data were obtained from Scofield et al. 2011.

11.6 Discussion

The LCR periphyton communities are productive, diverse and variable. The causes for this desirable condition are discussed in the following sections.

Periphyton sampling was focused on the most productive area of the river - the permanently wetted, shallow substrates in LCR Reach 2, all in the photic zone from the water's edge to depths of 5 - 6 m. Sites that experienced frequent dewatering of their shallow and mid-shallow sections had lower productivity and diversity. The transect depth where peak biomass occurred varied with season and sample site.

11.6.1 Periphyton Community Composition

Species diversity indices indicate that LCR periphyton biodiversity is stable and moderate compared to other large rivers (Table A22). Periphyton community compositions exhibited significant differences between years and seasons, and to a lesser extent between depth and site. Some of this variance may relate to flows and LCR operating regime, while some is likely attributable to variable nutrient inputs and reservoir phytoplankton donations, together with weather effects.

Like in all large rivers, diatoms dominated the LCR periphyton every year, with variable contributions made by soft-bodied algae such as flagellates, filamentous greens and cyanobacteria. As other research repeatedly shows, the high-profile filamentous taxa are readily torn off in velocities exceeding 0.2 m/s (Hart et al. 2013). The nuisance diatom Didymo was detected at all LCR sample sites and was most prevalent in winter at sites where rocky substrates with cool, clear, moderate flows favoured its growth (Bergey et al. 2009; Bothwell et al., 2009; Shelby 2006; Bunn and Arthington 2002). Species richness was lowest in the fall, particularly at the shallow sites that experienced dewatering and at deep sites where light penetration was low.

The periphyton community composition of winter was distinct from the periphyton community composition of summer and fall. The ALR donates variable amounts of phytoplankton in each season, and in the low flow seasons, it contributes most to sites closest to the HLK dam. Reservoir periphyton contributions are significant to LCR and have been observed in other river systems immediately downstream of a reservoir (Truelson and Warrington, 1994; Bonnett *et al.* 2009). Differences detected in the winter periphyton community composition are likely a result of increased Didymo growth.

In summary, hydraulic conditions in general and managed flows in particular can influence periphyton community structure (Hart et al. 2013), however, there are numerous other drivers such as nutrients, phytoplankton donations and weather that also play a role (Hart et al. 2013).

11.6.2 Periphyton Production – Standing Crop

The LCR periphyton communities are productive, diverse and variable. Most production metrics place LCR in the typical to productive range compared to other large rivers (Table A5-2). The most productive sites were S1 and S7 and they were both directly affected by reservoirs that each had fertilization programs.

The only consistently depositional site in all flows, S6, had the lowest periphyton productivity across all seasons compared to erosional sites, despite nutrient inputs from the municipal outfall. Stable erosional substrate provide long-term habitat for the larger, slow-growing periphyton and this contributes to standing crop productivity (Morley et al. 2008). Additionally, periphyton productivity was shaded by a macrophyte overstory at Site 6.

The transect depth where peak biomass occurred varied with season and sample site. The transect position with the greatest periphyton abundance, biovolume and chl-a was mid-shallow-MS to mid-M in winter but shifted to mid-deep MD in fall. A combination of water velocity and light intensity at the substrates is probably responsible for this observation. This is another line of evidence confirming that flow regime affects periphyton productivity.

Table A22. Summary of typical LCR periphyton metrics from 2008 to 2016, with comparisons to oligotrophic, typical, productive large rivers and MCR.

Metric	**Oligotrophic or stressed rivers**	**Typical large rivers**	**Eutrophic or productive rivers**	**MCR**	**LCR (median)**
Number of taxa (live & dead)	<20 – 40	25 - 60	variable	5 - 52	8 – 75 (32)
Chlorophyll-a $\mu g/cm^2$	<2	2 - 5	>5 – 10 (30+)	0.04 – 4.1	0.01 – 55 (3.6)
Algae density cells/cm^2	<0.2 x10^6	1 - 4 x10^6	>10 x10^6	<0.02 – 1.5 x10^6	0.03– 4.1x10^6 (0.8x10^6)
Algae biovolume cm^3/m^2	<0.5	0.5 – 5	20 - 80	0.03 - 10	0.1 – 41 (3.5)
Diatom density frustules/cm^2	<0.15 x10^6	1 - 2 x10^6	>20 x10^6	<0.01 – 0.6 x10^6	0.06 – 6.91 x10^6 (0.56)
Biomass –AFDW mg/cm^2	<0.5	0.5 - 2	>3	0.12 – 4.8	0.04 – 20.3 (0.53)
Biomass –dry wt mg/cm^2	<1	1 – 5	>10	0.7 – 80	0.18 - 429
Organic matter (% of dry wt)		4 – 7%		1 – 10%	0.38 – 44.6 %
Bacteria sed. HTPC CFU/cm^2	<4 -10 x10^6	0.4 – 50 ×10^6	>50×10^6 – >10^{10}	0.2 – 5 x10^6	1.5 - >5 x 10^6
Fungal count CFU/cm^2	<50	50 – 200	>200	<25 – 600	8 - 1830
Accrual chl-a $\mu g/cm^2/d$	<0.1	0.1 – 0.6	>0.6	0.001 - 0.1 S 0.005 - 0.38 D	0.009 – 0.44 S 0.015 – 0.51 D

Comparison data obtained from Flinders and Hart 2009; Biggs1996; Peterson and Porter 2000; Freese *et al.* 2006; Durr and Thomason 2009; Romani 2010; Biggs and Close 2006.

11.6.3 Periphyton Accrual

Although LCR periphyton standing crop metrics were variable between seasons and years, the chl-a weekly time series samples indicated that accrual reaches peak biomass in 6-7 weeks in summer, greater than 8 weeks in fall and greater than 10 -12 weeks in winter. When samplers were deployed for longer than these periods, a combination of sloughing due to flow change, grazing, and shading by surface algae layers and bacterial decomposition of algae cells deep in the periphyton biofilm, all acted to limit the standing crop of periphyton in LCR.

LCR accrual rates were apparently slower in winter than in summer and fall, suggestive of temperature effects where, for example, large filamentous taxa like Didymo and Stigeoclonium are disadvantaged at water temperatures below 10^{o}C. Slower accrual rates in cool water are widely encountered (He 2010; Vermaat and Hootsmans 1994).

11.6.4 Influence of Managed Flows on LCR Periphyton Community Structure and Productivity

This appendix answers ***Management Question 1.*** *What is the composition, abundance, and biomass of epilithic algae (periphyton) in LCR? 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 epilithic algae?*

No data were collected during the unmanaged flow periods prior to the early 1990's, preventing a before/after analysis. Instead, reference years were selected that had flows similar to the unmanaged period. We then used the lines of evidence approach.

Periphyton in LCR showed significant variations in production and community structure between seasons and years. Many factors that influenced periphyton production gradients are related to LCR flows contributed by reservoir releases. Our field observations agree well with statistical analyses - flow variability and velocity are the important factors influencing periphyton production in LCR. These results indicate a direct link between productivity and operations. Additionally, flows affected turbidity and nutrient concentrations with consequences for periphyton community structure and standing crop (see Appendix 5).

Although river discharge clearly influenced the LCR periphyton community, the influences of managed fish flows (MWF, RBT and FFF) are difficult to discern. Each managed flow period and relevant hypothesis is considered separately in the following sections.

MWF Flow (winter)

Lower temperatures of 4 – 6^{o}C and reduced light intensity coupled with shorter day length apparently exerted less influence than the benefits of stable winter flows because winter samplers showed higher overall periphyton production than other flow periods; however, the time to achieve peak biomass was longer. This implied a slower overall growth rate. Cool winter water temperatures will restrict growth of most green algae and some cyanobacteria, but not diatoms or most flagellates (Wetzel 2001), explaining the very low abundance of filamentous green algae in winter samples and the prevalence of low-light tolerant cyanobacteria, and diatoms including Didymo. A more diverse periphyton community developed in 2013 relative to all other MWF flows, indicating that periphyton community diversity decreased with managed flows.

Based on the results to date, we reject hypothesis HO_{2Aeco} that MWF flows do not increase total accrual of periphyton or their biomass. The lines of evidence to support this rejection of hypothesis HO_{2Aeco} include:

- Comparison of winter 2013 productivity metrics, when flows were similar to unregulated flows, with winter 2014, 2016 and 2018, revealed variability among years. There was higher chl-a in 2016 compared to 2013, but lower biovolume in all subsequent years (and significantly so in 2014) compared to the reference year 2013.
- In this study, lower stable flows during winter with MWF managed flows were associated with higher biomass, particularly of the high-profile guild, and slower growth rates.
- The transect depth where peak biomass occurred was Mid-shallow to Mid in winter but shifted to Mid-deep in fall, indicating that flow regime affects periphyton productivity, and shifts the depth at which peak biomass occurs.

RBT Flows (summer)

The lowest overall periphyton production and diversity were observed during the summer when freshet was occurring. Shear and scour of periphyton from higher velocities during high flow periods are likely the cause of this observation[3]. Reduced periphyton growth following high flow events is frequently observed in other river systems (Blinn *et al.* 1995, Biggs 1996, Bunn and Arthington 2002). Specifically, high-profile filamentous green taxa and Didymo masses can be dislodged with small increases in velocity above 0.2 m/sec, while tightly attached low-profile diatoms require increased shear stresses to experience the same scour (Biggs 1996).

High freshet flows dominate the RBT flow period, both in scale and apparently in effect on periphyton biomass metrics. Generally, sites and years with higher velocities had lower periphyton production. Averaged hydrographs show managed flows are more consistent.

We reject the null hypothesis HO_{2Beco} that RBT flows do not increase total biomass accrual of periphyton in LCR. The lines of evidence to support this rejection of the null hypothesis HO_{2Beco} include:

- During the summer RBT flow period, both peak velocities and flow variability are reduced compared to unmanaged summer flows. Over the managed RBT periods, periphyton community shifts occurred that were likely induced by flows. For example, the percent high-profile guild was significantly higher in years with moderate RBT flows compared to the extreme flows in 2012. These results indicate that the greater peak flows and wider flow variability that characterized unmanaged flows would reduce periphyton growth compared to managed RBT flows.
- More substrate dewatering occurred prior to RBT flow management, with flows dropping below 600 m^3/sec in late April and in June (Figure 4-6). With management, flows less than 600 m^3/sec were eliminated improving periphyton productivity at shallow sites which should result in greater overall periphyton productivity.

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

Fall Fluctuating Flows

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

Based on the data to date, we reject the null hypothesis HO_{2Ceco} that FFF do not increase total biomass accrual of periphyton in LCR. The managed flows had less flow variability and less substrate dewatering, both factors associated with greater periphyton productivity in LCR. The lines of evidence to support this rejection of the null hypothesis HO_{2Ceco} include:

- Fall periphyton production (total biovolume and chl-a) data indicate that FFF flow variability decreased total biomass accrual. Thus, decreased flow variability with managed FFF flows compared to unmanaged fall flows should allow greater periphyton biovolume and chl-a.
- Substrate dewatering was less severe and sustained with FFF managed flows than it was under unmanaged fall flows and should result in greater overall periphyton productivity. The cost of dewatered substrates to periphyton productivity is well-known in MCR (Larratt et al., 2017). In LCR, substrate dewatering was greatest during FFF (late Sept and Oct) and resulted in the expected loss of periphyton productivity in the shallows. Minimizing substrate dewatering in the fall with managed FFF flows maintains greater periphyton productivity in the FFF sampling period.

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12.0 APPENDIX 7. Ecological Productivity Monitoring - Management Question #2

12.1 Introduction

This appendix addresses the Ecological Productivity Management Question #2 and associated hypotheses.

MQ#2 *What is the composition, abundance, and biomass of benthic invertebrates in LCR? 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?*

HO_{2eco}: *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_{2Aeco}: *Continued implementation of MWF does not affect the biomass, abundance and composition of benthic invertebrates in LCR.*

HO_{2Beco}: *Continued implementation of RBT flows does not affect the biomass, abundance and composition of benthic invertebrates in LCR.*

HO_{2Ceco}: *Continued fluctuations of flow during the fall do not affect the biomass, abundance and composition of benthic invertebrates in LCR.*

12.2 Methods

12.2.1 Data Collection

Benthic invertebrate productivity was determined with the use of artificial substrates placed at seven sampling sites (S1-S7) within Reach 2 (Figure A39) during three seasons. Invertebrate sampling in later years of the study differed from Years 1-3, in that all sampling locations were in Reach 2. The reader should refer to annual reports for additional sampling locations and methodologies used during the first three years of the study (TG Eco-Logic 2009, 2010 and Scofield et al. 2011).

In 2012, the transect depths sampled at each site were increased from three depths to five. Previously, depths were referred to as shallow [S], mid [M], or deep [D]. The five depths sampled since 2012 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 A23), with the intermediate depths providing more data for statistical evaluation.

Table A23. Naming Convention of Sampling Depths and Corresponding Depth Strata

Depth Label	Depth Name	Depth Strata (m)	Years sampled
D	Deep	>5.5	2008 – 2010, 2012, 2014, 2016, 2018
MD	Moderately deep	4 – 5.5	2012, 2014, 2016, 2018
M	Mid	2.5 – 4	2008 – 2010, 2012, 2014, 2016, 2018
MS	Moderately shallow	1 – 2.5	2012, 2014, 2016, 2018
S	Shallow	<1	2008 – 2010, 2012, 2014, 2016, 2018

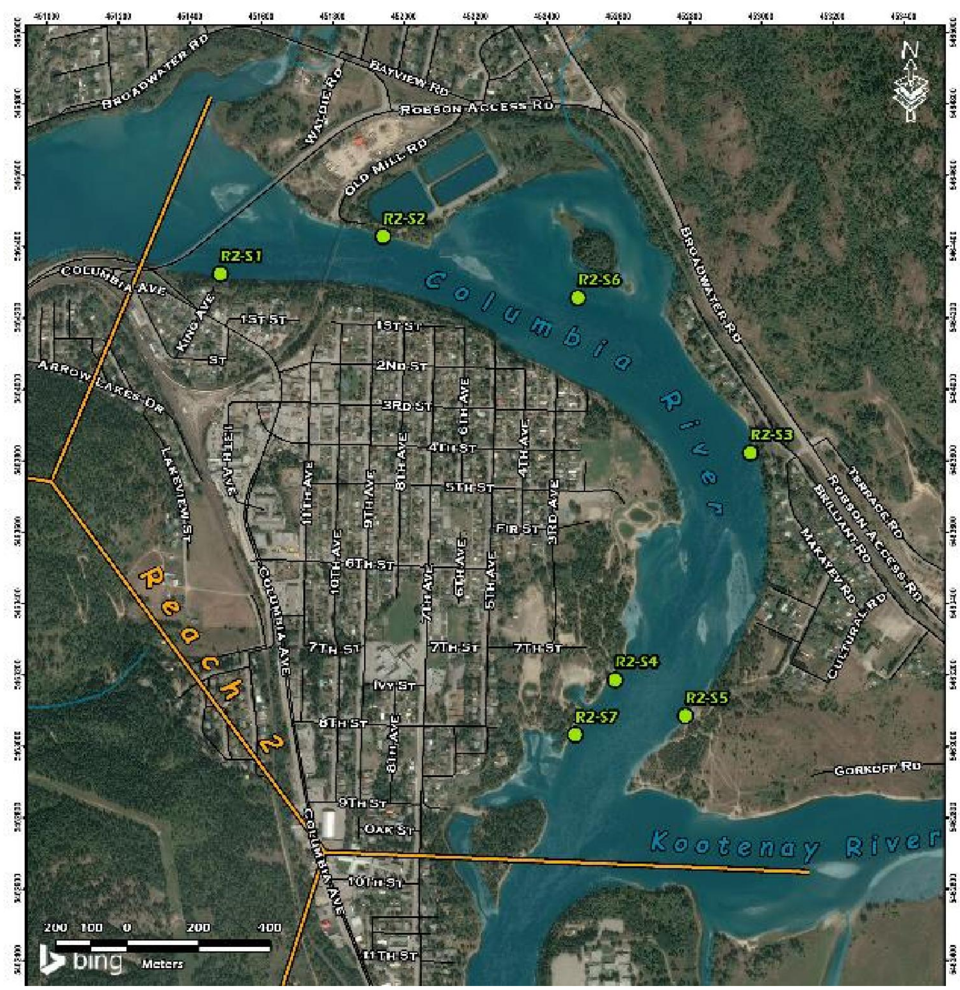

Figure A39. Reach 2 Benthic Productivity Sampling Locations.

In 2018, a single artificial sampler apparatus design was used for all seasons over a 10-week sampling duration (Figure 3-1). The winter samplers were deployed from January 9th through March 20th. The sampling session was designed to coincide with the MWF flow period. The summer sampling period occurred from June 1st through August 9th and the fall sampling period occurred from August 9th through October 17th. The winter and fall sampling sessions entirely overlap with MWF and FFF flows, while only the first month of the summer deployment overlaps with the RBT flow period. Table A24 provides deployment dates, sampling numbers and equipment/sample recovery rates.

Table A24. Benthic Invertebrate Sampler Deployment and Recovery Rates in 2018.

Season	Reach	Site	Invertebrate Basket Samplers: # Deployed	Invertebrate Basket Samplers: # Retrieved (% Recovery)
Winter (Jan 9 - Mar 20) 10 weeks	2	Site 1 (S1)	5	5 (100)
		Site 2 (S2)	5	5 (100)
		Site 3 (S3)	5	5 (100)
		Site 4 (S4)	5	5 (100)
		Site 5 (S5)	5	4 (80)
		Site 6 (S6)	5	5 (100)
		Site 7 (S7)	5	5 (100)
Winter Totals			**35**	**34 (97)**
Summer (Jun 1 - Aug 9) 10 weeks	2	Site 1 (S1)	5	4 (80)
		Site 2 (S2)	5	5 (100)
		Site 3 (S3)	5	4 (80)
		Site 4 (S4)	5	5 (100)
		Site 5 (S5)	5	4 (80)
		Site 6 (S6)	5	5 (100)
		Site 7 (S7)	5	5 (100)
Summer Totals			**35**	**32 (91)**
Fall (Aug 9 - Oct 17) 10 weeks	2	Site 1 (S1)	5	5 (100)
		Site 2 (S2)	5	5 (100)
		Site 3 (S3)	5	3 (60)
		Site 4 (S4)	5	4 (80)
		Site 5 (S5)	5	5 (100)
		Site 6 (S6)	5	5 (100)
		Site 7 (S7)	5	5 (100)
Fall Totals			**35**	**32 (91)**
2018 Totals			**105**	**98 (93)**

Lost samplers were mostly due to shallow samplers being pulled to shore by bystanders.

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

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

12.2.2 Laboratory Processing

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

The data sets used to analyze benthic invertebrate Ecological Management Question #2 are provided in Table A25, below.

12.3 Datasets

Table A25. Datasets used in the analysis of ecological productivity management question #2.

Name	Data Source/Description	Years Obtained
Light / Water Temp	Data collected at each productivity sampler during each deployment session	2008 – 2010, 2012, 2013 (winter only), 2014, 2016, 2018
Benthic Invertebrates	Data collected at each productivity sampler during each deployment session. Data produced in the laboratory included abundance, biomass, and associated metrics. Additional metrics described in Table A26 were calculated.	2008 – 2010, 2012, 2013 (winter only), 2014, 2016, 2018
Velocity	Data collected at each productivity sampler twice per deployment period	2009 – 2010, 2012, 2013 (winter only), 2014, 2016, 2018
Substrates	Substrate percentage at each deployment site estimated during each deployment period	2009 – 2010, 2012, 2013 (winter only), 2014, 2016, 2018

12.4 Analysis

12.4.1 Benthic Invertebrate Community Analysis

Non-metric multidimensional scaling (NMDS) was used to explore variation in benthic community composition at the family level. The Bray-Curtis dissimilarity index was used, as it is sensitive to the variation of species that have smaller abundances (Clarke and Warwick 1998). To visually explore differences in community compositions, the NMDS scores for samples collected between 2008 and 2018 were plotted using R package ggplot2 (Wickham 2009). A permutational multivariate analysis of variance (PERMANOVA) was used to determine if there were significant differences in community compositions according to depth, site, season and year. The amount of variability in community composition was determined by calculating the partial R^2 from a permutational MANOVA. Both NMDS and permutational MANOVAs do not make assumptions of the variable distributions and relationships (Anderson 2001; Clarke *et al.* 2006). The NMDS analysis and permutational MANOVA used R package vegan version 2.3-5 (Oksanen *et al.* 2016). The NMDS analysis was performed with rare taxa included and excluded and both results were very similar. Rare taxa were defined as taxa that represented less than 5% of the total samples. The results presented are with rare taxa excluded. To identify taxonomic differences between samples, taxa were related to the community differences by fitting them to the ordination plot as factors using Envfit (Oksanen et al. 2017).

To better understand the invertebrate productivity and community composition, the metrics in Table A26 were calculated. In addition, the percent abundance of the top three dominant taxa in each unique season, site and year combination was calculated.

Table A26. **Responses for Benthic Invertebrates.**

Variable	Description
Total Abundance	Total Abundance across all species
Total Biomass	Total Biomass across all species
Effective Number of Species	A measure of community diversity that is the e^s. S= Shannon-Wiener index.
Percent EPT	The percentage of Ephemeroptera, Plecoptera, and Trichoptera based on biomass
Species Richness	Number of unique species
Percent Chironomidae	The percentage of Chironomids based on biomass
Fish Food Biomass (Good Forage)	Calculated by summing the biomasses of Ephemeroptera, Trichoptera and Plecoptera, and Dipteran species, all considered good fish forage

12.4.2 Benthic Invertebrate Production

Linear mixed effects models were used for the MWF flow period to compare annual variations in benthic invertebrate productivity and community composition. Data from the shallow (S) samplers was excluded from the analysis, because in some years they were prone to exposure due to low flows or tampering. Total abundance and total biomass were log10 transformed to reduce heteroscedasticity in model residuals, whereas effective number of species did not require a transformation. Velocity was used as a fixed effect in all linear mixed effects models (Table A27). The benthic invertebrate sampling design does not include true replicates. However, there is pseudo-replication among benthic invertebrate samples. The level of pseudo-replication is difficult to determine but it is expected that pseudo-replication occurs at the site level. In some cases, pseudo-replication may also occur at the site and year level within a given season.

The correct selection of a random effect is required to ensure the linear mixed effects model does not violate the assumption of non-independent observations. Separate models for the MWF flow period were fit with site as a random effect and site and year combination as a random effect. If the model that included site and year as a random effect had a lower AIC than the model that included site as a random effect it was selected for the final mixed effects model. For total abundance model site was used as the random effect. The random effect for the effective number of species and total biomass models was the combination of site and year. The 95% confidence intervals for the fixed coefficient of year and velocity were calculated and plotted using the R package jtools version 2.0.1 (Long 2019).

Table A27. Explanatory Variable for both Benthic Invertebrates

Variable	Description
Velocity	Velocity was measured on the day of deployment and the day of retrieval. The average of these two values was used in the analysis.

12.5 Results

12.5.1 2018 Benthic Invertebrate Community Compared to Previous Years

Rock basket substrates to determine the benthic invertebrate community in LCR were deployed during the winter, summer and fall of 2018. During the three sampling sessions, 93% of deployed rock baskets were recovered and analyzed (Table A28). Most of the loss that occurred was at the shallow depth and was a result of the samplers being pulled to shore during lower flows and left on the riverbank exposed.

Table A28. Rock Basket Recovery by Season in 2018. Fractions indicate the number of substrates recovered over the number of substrates deployed.

Season	Recovery Rate	Percent Retrieved
Winter	34/35	97
Summer	32/35	91
Fall	32/35	91
Total	98/105	93

As with previous years, LCR had an abundant and diverse community of benthic macroinvertebrates in 2018. The 2018 benthic invertebrate data varied by season. The highest mean abundance (#/basket) ± SD occurred in the summer with 9,385 ± 8,067 organisms per basket, followed by fall (4,156 ± 3,439) and winter (2,324 ± 1,864) (Figure 4-14). Similar to 2016, fall samples had the highest biomass, followed by summer, while winter was substantially lower (Figure 4-15). The winter 2018 biomass data had a similar range to the 2012 and 2016 winter biomass data. Both the 2018 abundance and biomass data fell within the range of previous sampling periods. However, summer 2018 data was similar to 2016 with higher productivity compared to previous sampling years (Figure 4-14 and Figure 4-15).

The mean species richness was very similar across the three seasons, ranging from 23 ± 8 in the summer, to 21 ± 6 in the fall and 19 ± 6 in the winter (data not shown). The effective number of species refers to the number of equally abundant species needed to obtain the same mean proportional species abundance as that observed in a dataset. It is a stable and sensitive similarity measure which is easier to interpret than other indices such as Shannon-Wiener (Jost 2006). Figure A40 depicts the effective number of species for each season, across all years of sampling. Interestingly, winter 2014 had the lowest effective number of species, even though it had an unusually higher biomass compared to other winter sampling events (Figure 4-15).

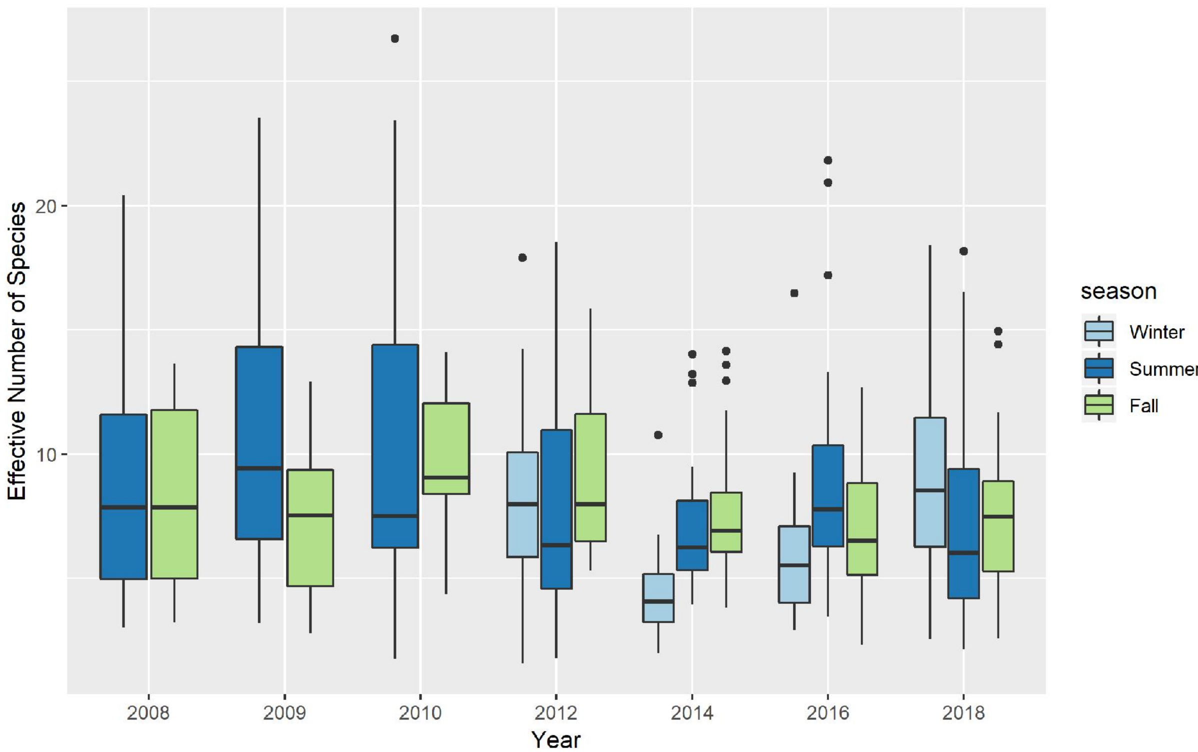

Figure A40. Effective number of species across all years and seasons.

Dominant taxa in summer and fall 2018 included Hydropsychidae (net-spinning caddisflies) and Chironomidae (Figure A41). In the winter, Simuliidae (black fly) was the dominant taxa across three of the seven sampler depths. Chironomidae also remained a dominant winter taxon. Dominant invertebrates in summer and fall 2018 were very similar to those sampled in previous years, while the dominant winter 2018 taxa were most similar to 2013 and 2014 but differed from 2016. Simuliidae presence was limited in winter 2016, particularly at sites S1 – S3 (Figure A41).

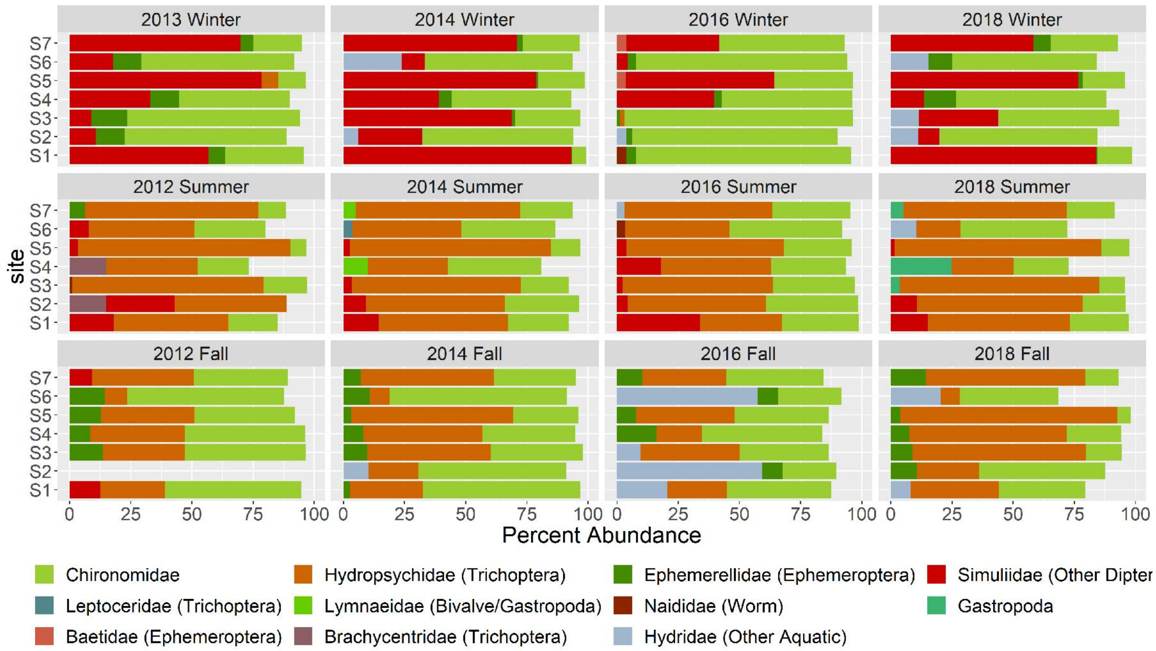

Figure A41. Dominant taxa at the family level for each year when sampling occurred during winter, summer and fall. Note: winter samples in 2013 should be compared to summer/fall of 2012.

Community analyses of the invertebrate data was also completed at the family level. The NMDS stress index was 0.23, which indicates the two NMDS axes partially explain the invertebrate community composition. A permutational MANOVA indicated that season explained the most variation (R^2=0.11) in invertebrate community compositions and was significant (F=31.6, p<0.001). The separation of invertebrate community compositions by season was especially evident in winter which was distinct from fall and summer (Figure A42). Year, depth and site were also significant, but these parameters explained less variation in the invertebrate community composition (R^2=0.02 – 0.08) (Table A29).

Table A29. PERMANOVA results for benthic invertebrates at Family level.

group	R_stat	Fstat	p_val
year	0.030	14.520	<0.001
depth	0.020	2.060	0.005
site	0.080	7.550	<0.001
season	0.110	31.630	<0.001

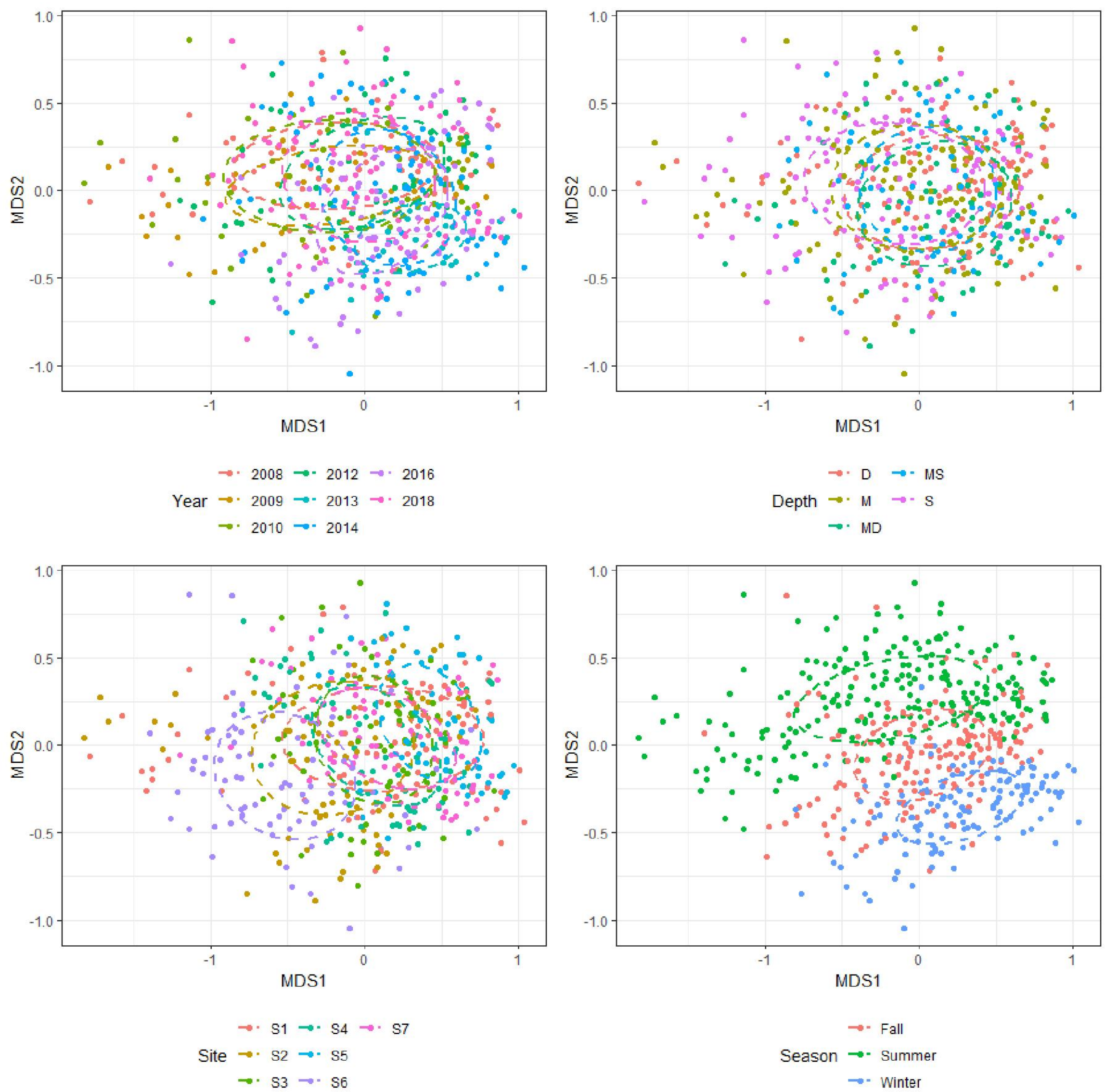

Figure A42. NMDS plots of LCR benthic invertebrates (at Family level) by year, depth, site and season. The NMDS used a Bray-Curtis dissimilarity index and had a stress index of 0.23. Ellipses are calculated based on 95% Confidence Interval of the NMDS scores for each group.

The benthic invertebrate families that played the largest role in the seasonal community separation was Hydropsychidae (Table A30). Hydropsychidae was positively associated with Axis 1, which means summer on average has the most Hydropsychidae followed by fall and summer. The dominant taxa analysis confirms that Hydropsychidae was the dominant species in summer, with smaller abundances in the fall and it was essentially absent during the winter (Figure A41).

Table A30. Taxa scores for NMDS axes with p-value and R^2.

Species	NMDS1	NMDS2	pval	r2
Hydropsychidae	0.18	0.5	0	0.29
Asellidae	-0.44	-0.07	0	0.2
Crangonyctidae	-0.44	0	0	0.19
Simuliidae	0.38	-0.15	0	0.17
Glossiphoniidae	-0.37	0.01	0	0.14
Lymnaeidae	-0.12	0.34	0	0.13
Hydroptilidae	-0.05	-0.34	0	0.12
Baetidae	0.32	-0.12	0	0.11
Chironomidae	0.29	0.17	0	0.11
Planorbidae	-0.21	0.26	0	0.11
Erpobdellidae	-0.32	-0.02	0	0.1
Gastropoda	-0.11	0.3	0	0.1

As discussed in Section 4, the MWF flows in 2013 most resembled pre-MWF flows (Figure 4-2 and Figure 4-5). Benthic invertebrate productivity measures including abundance, biomass and effective number of species were used as metrics to assess invertebrate productivity in linear mixed effects models. Both abundance and biomass had a positive association with velocity during the MWF flow period, while the effective number of species was negatively associated with velocity (Figure A43). During the MWF period, biomass was significantly higher in 2014 compared to 2013. Also, 2014 had a significantly lower number of effective species compared to 2013 (Figure A44).

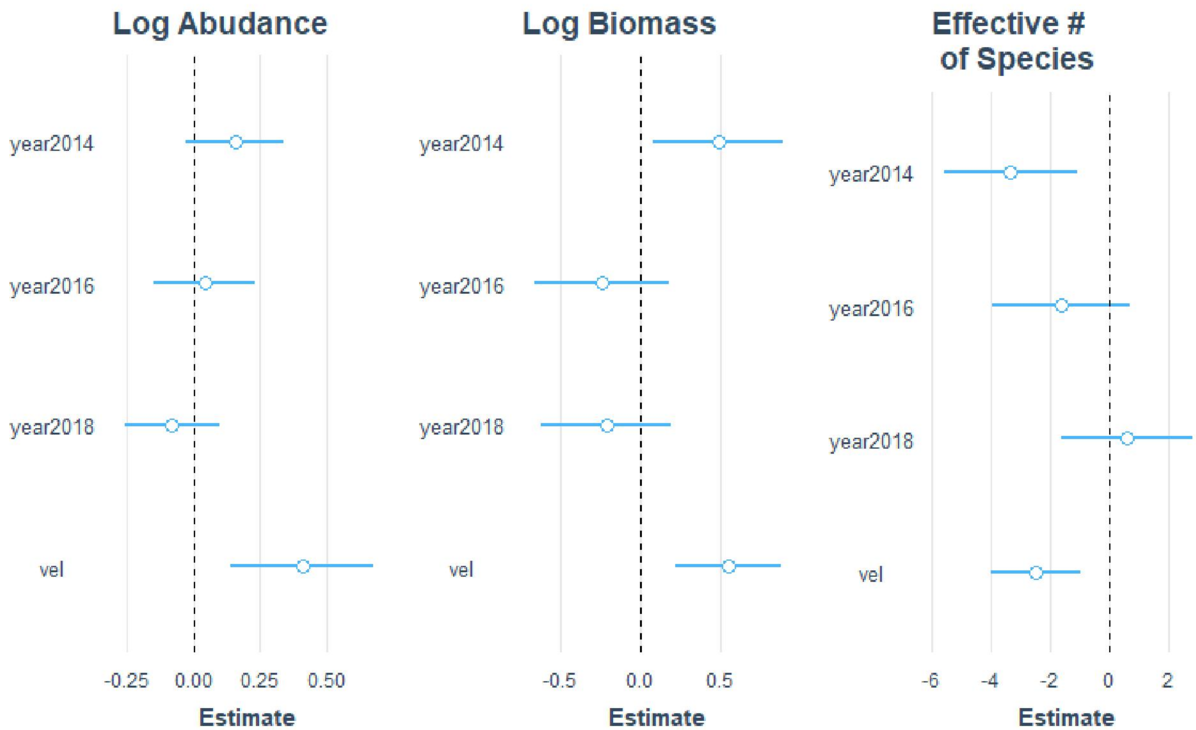

Figure A43. Benthic Invertebrate Productivity and Composition for the MWF flow period, fixed effects confidence intervals for velocity and year compared to reference year 2013.

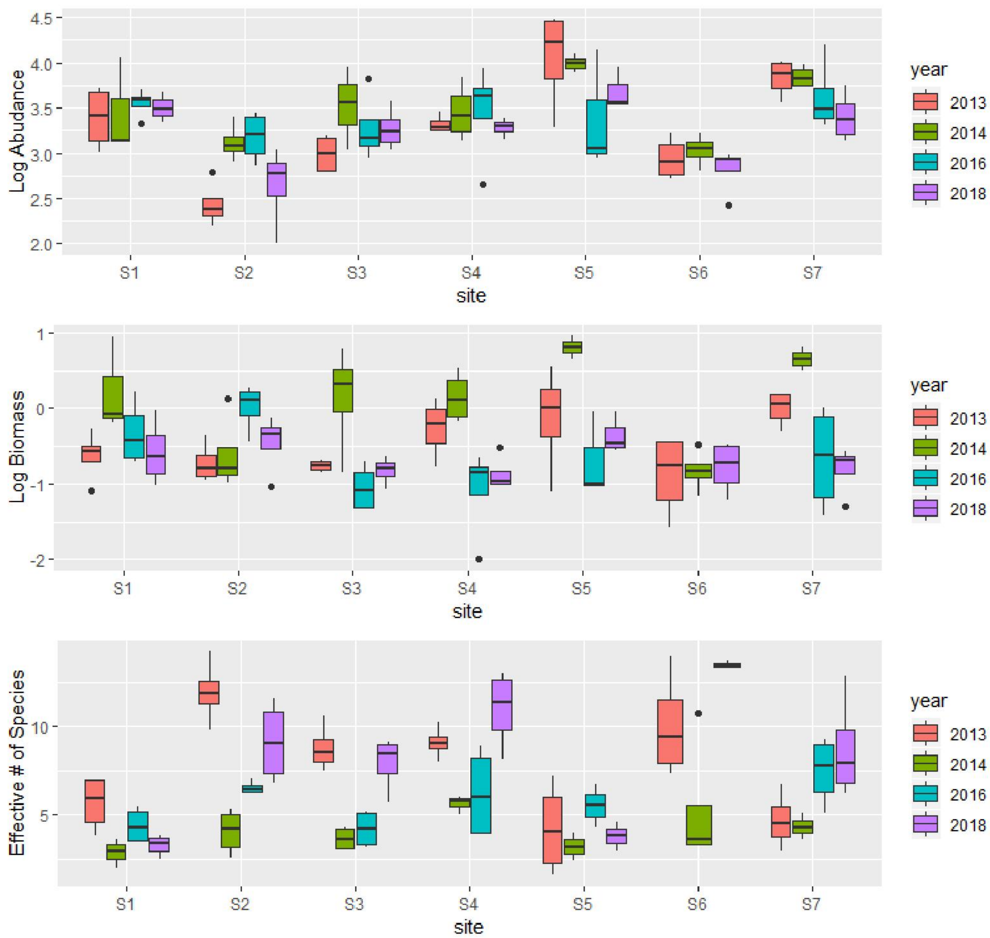

Figure A44. Benthic invertebrate abundance, biomass and effective number of species for the MWF flow period compared across years and sites.

12.6 Discussion

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

The managed flow periods have been implemented in LCR long enough that resulting shifts in the benthic invertebrate community have likely stabilized (Poff and Zimmerman 2010). Seven years of benthic invertebrate data were collected between 2008 and 2018, but no data was collected prior to the implementation of managed flows.

Flow is widely recognized as a major determinant of both physical habitat and biotic composition in rivers, and that aquatic species have life history strategies that respond to natural flow regimes (Bunn and Arthington 2002). Likewise, the effects of large impoundments on river ecology has also been well documented (Bunn and Arthington 2002; Konrad et al. 2011). The focus of this study was to understand if the managed fish flows have influenced the benthic invertebrate community, beyond what is typical given LCR's impoundment and the deviation from a natural system. In both cases, the MWF and RBT flow regimes are designed to reduce daily and seasonal flow variability.

Despite some similarities of the annual LCR hydrograph to a natural system, hydrologic differences do exist. In other river systems, flow regulation has been shown to favour less sensitive invertebrate species such as orthoclad chironomids (Poff and Zimmerman 2010; Munn and Brusven 1991), and Simuliidae (Ellis et al. 2016). Both of these taxa are dominant invertebrates in LCR. Chironomidae and Simuliidae comprise more than 80, 50 and 35 percent of the invertebrate abundance in winter, fall and summer, respectively. An increased predominance of filter-feeding benthic invertebrates has also been documented in regulated river systems (Kjaerstad et al. 2017), and LCR has high relative abundances of net-spinning caddisflies of the family Hydropsychidae during the fall and summer. Also, there is a lower abundance of Ephemeroptera (mayflies) which are sensitive to changes in flow (Szczerkowska-Majchrzak *et al.* 2014; Kennedy *et al.* 2016). Thus, in these features, the LCR benthic invertebrate community is typical of a regulated river system. However, given its ranking as a diverse and productive system, regulation has not resulted in excessive impairment of benthic invertebrate productivity.

Coupled with the effects of regulation on the invertebrate community, other variables such as nutrient additions through the ALR fertilization program, industrial effluents (Zellstoff Celgar), municipal effluents, and nuisance Didymo all influence the overall distribution, abundance, and diversity of the LCR benthic invertebrate community. This makes it difficult to separate the effects of a given flow regime from natural, annual and seasonal variation, and from variation originating from the influences of other ongoing factors (Bunn and Arthington 2002). Thus, distinguishing the effects of flow regulation, and more specifically the effects of managed fish flows, from other stressors and inherent natural patterns on the benthic community, has not been accomplished with statistical rigor. Rather, we relied on lines of evidence, our understanding of the river system, statistical modeling and the literature to support or reject the effects of MWF, RBT and FFF flow periods on the benthic invertebrate community.

The LCR invertebrate community composition in winter during MWF flows was distinct from the fall and summer sampling periods. Percent EPT was consistently lower in winter compared to other sampling seasons, while Dipteran Simuliidae maintained a higher abundance. Tonkin et al. (2017) demonstrated that benthic invertebrate communities in highly seasonal environments exhibit strong fluctuations in community structure with turnover from one unique community type to another across seasons. Seasonal aquatic communities can be taxonomically distinct; often characterized by species that differ greatly in thermal tolerance and trophic position (Bêche et al. 2006, Bonada and Resh 2013).

As previously discussed, the flows during 2013 most closely resembled pre-managed flow conditions with higher daily and seasonal variability. When the 2013 benthic invertebrate community, was compared to other sampled years, 2014 was the only year that showed significant differences in biomass, biomass of EPT+D and effective number of species. Winter 2014 had a less diverse benthic invertebrate community that had higher biomass and availability of fish food organisms compared to winter 2013. The higher biomass of EPT+D and lower effective species number was a result of a dominance of Simuliidae in winter 2014. The higher biomass of Simuliidae in winter 2014 is thought to be a result of smoother rock surfaces which was caused by lower periphyton biovolumes. Simuliidae have shown preferences to smooth rock with less algae growth (Mackay, 1992).

Although the winter of 2013 had higher daily and seasonal variability in flow, there were no obvious shifts in benthic invertebrate abundance or biomass. The benthic invertebrate literature indicates that flow fluctuations without substantial hydropeaking result in unaltered functional feeding groups across lateral transects (Kjaerstad et al. 2017). In other words, if the river substrates are submerged, benthic invertebrate indices are not impacted. The benthic invertebrate samples collected as part of this program provide an estimate of the productivity for the permanently submerged areas in LCR.

The higher flows at the start of the MWF flow period before the management of MWF flows would have resulted in a larger wetted habitat area. The larger wetted area would have then resulted in a greater invertebrate production at the beginning of the MWF flow period, followed by a substantial drop in flows, and varial zone dewatering. The differences in wetted habitat area from pre and post MWF flows have not been quantified but the comparison of predicted elevation differences suggests there is a substantial difference in wetted habitat area. The area of potentially lost habitat will be explored in future years where the bathymetry of LCR near the Norn's Creek fan will be surveyed and the areas of submerged habitat will be modelled under different flow regimes (BC Hydro 2019).

Interestingly, the dominant invertebrate taxa in LCR during winter include chironomids and Simuliidae. These taxa are tolerant to desiccation and are commonly found in the upper varial zone of rivers (Jones 2013). Simuliidae are particularly resistant to cold temperatures and long periods of dewatering and exposure (Davies 2017). They are also capable of rapidly recolonizing recently exposed substrates through a combination of morphological adaptations (small body size, silk hooks to anchor on substrates) and behaviour (looping movements locate new substrates under high flow conditions) (Kjaerstad et al. 2017, Zhang et al., 1998).

So even though the invertebrate community is specialized for the recolonization of exposed areas, and other lines of evidence suggests that the submerged invertebrate community is highly tolerant of changes in flow, the overall smaller area of submerged habitat that results from managed MWF flows results in less benthic productivity compared to pre-managed flows when a greater area of habitat was submerged, albeit for an abbreviated time.

Given that the managed MWF flows result in less submerged habitat, ***we reject the management hypotheses HO_{2eco} which states that the continued implementation of MWF flows during winter, does not affect the biomass, abundance and composition of benthic invertebrates in LCR.*** We believe that these measures are less due to a reduction in overall wetted habitat.

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. Prior to the RBT managed flows, the discharge from HLK exhibited two prolonged drops in flow, one in May and one in June with drops of 200 and 400 m/s^3, respectively. These drops lasted up to 4 weeks and would have resulted in substantial areal losses of submerged and productive habitat.

By stabilizing the increasing freshet flows, the continued implementation of RBT flows results in a more stable benthic invertebrate community that encompasses a larger area of the river due to more submerged riverbed. ***We therefore reject the hypothesis that the continued implementation of RBT flows does not affect the biomass, abundance or composition of benthic invertebrates in LCR.***

The increased wetted habitat area resulting from the managed RBT flows provides favourable growth conditions for periphyton and benthic invertebrates alike. Hydropsychidae (net-spinning caddisflies, Trichoptera), the most abundant invertebrate taxa during the summer, prefer high water velocities and substrates with substantial periphyton coverage (McKay 1992). These conditions are promoted by stable RBT flows. Additionally, Hydropsychidae was the dominant family in all fall 2012 and 2014 RBT fish stomach samples. Hydropsychidae are likely the primary food source during the summer when their abundances are greatest.

The increased invertebrate productivity that results from RBT managed flows also provides a benefit during the FFF period. Taxa such as Hydropsychidae and Ephemerellidae have larvae that require periphyton and warmer water temperatures (Mackay 1979). Ephemerellidae larvae require longer periods for growth and as a result, emerge in fall (Raddum et al. 2008). These larvae become established during the RBT flow period and emerge during late summer and fall, therefore contributing to invertebrate productivity and fish food availability during both RBT and FFF periods.

The stabilized and higher overall flows during the FFF period also result in a more stable benthic invertebrate community due to a greater area of submerged habitat. Prolonged dewatering results in losses to invertebrate abundance, biomass and diversity (Larratt et al. 2017; Kjaerstad et al. 2017). ***Based on this, we reject the hypothesis that the continued implementation of FFF does not affect the biomass, abundance or composition of benthic invertebrates in LCR.***

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13.0 APPENDIX 8. Ecological Productivity Monitoring - Management Question #3

13.1 Introduction

This appendix addresses the Ecological Productivity Management Question #3 and associated hypotheses.

MQ#3 *Are organisms that are used as food by juvenile and adult MWF and RBT in LCR supported by benthic production 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.*

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.*

13.2 Methods

13.2.1 Data Collection

Stomach contents were collected from 55 RBT and 65 MWF from various locations in LCR during the fall in 2012 and 2014 (Table A31). The fish were sampled as part of the CLBMON-45: Lower Columbia River Fish Population Indexing Survey (Golder Associates Ltd 2016). The stomach contents were obtained through gastric lavage, and contents were preserved in the field in 95% ethanol prior to shipment to the laboratory.

Table A31. Summary of RBT and MWF stomach contents obtained and analyzed.

Year	Rainbow Trout		Mountain Whitefish		
	Adult	Juvenile	Adult	Juvenile	
2012	13	2	23	2	
2014	32	8	29	11	
Totals	45	10	52	13	120

13.2.2 Laboratory Processing

The fixed stomach samples were analyzed for taxonomy and abundance by Cordillera Consulting in Summerland, B.C. Samples were sorted and identified to the genus-species level where possible. Benthic invertebrate identification followed standard procedures.

Macroinvertebrates 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 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). Detailed laboratory methods available upon request.

13.3 Datasets

Table A32. Datasets used in the analysis of ecological productivity management question #3.

Name/Description	Data Source	Years Obtained
Benthic Invertebrates	Data collected at each productivity sampler during each deployment session. Data produced in the laboratory included abundance, biomass, and associated metrics. Additional metrics described in Table A26 were calculated.	2008 – 2010, 2012, 2013(winter only), 2014, 2016, 2018
Fish Stomach	Fish stomachs obtained by Golder Associates Ltd as part of CLBMON-45 Fish Indexing Survey	2012, 2014

13.4 Analysis

The benthic invertebrate community composition of RBT and MWF stomach contents were analyzed at the family level. The percent abundance of the top five dominant taxa in each unique species and year combination was calculated. The fish stomachs were from juvenile and adult RBT and MWF caught in fall of 2012 and 2014. An NMDS using the Bray-Curtis dissimilarity index was conducted on the fish stomach community data. A PERMANOVA was used to determine if there were significant differences in community compositions according to year, species, or age (mature or juvenile).

To identify unique taxa in fish stomachs, taxa were related to the community differences by fitting them to the ordination plot as factors using Envfit (Oksanen *et al.* 2016). Only the taxa that were significant ($p<0.05$) and had r^2 greater than 0.1 were considered. These taxa describe the most observed variation between fish stomachs. Relative abundances of benthic invertebrate taxa were also calculated to identify dominant taxa. Dominant taxa could either be the most abundant taxa during the sampling period or the taxa that the fish prefer to consume.

To better understand how the availability of fish food impacts what invertebrate families are consumed by RBT and MWF, the top three taxa for each site were selected by calculating the relative abundance for all transects for each year and season combination.

The total biomass of Ephemeroptera, Plecoptera Trichoptera (EPT) and Diptera (D) was a response variable introduced in 2018 to test the availability of food for juvenile and adult MWF and RBT. Linear mixed effects models were used to determine if there were annual differences in the availability of fish food during the MWF, RBT and FFF flow periods. To determine annual differences in the total biomass of EPT+D, a linear mixed effects model was run separately for each flow period. The level of pseudo-replication for these models is expected to be site and year. However, the models were run with the combination of site and year and with site as a random effect. By comparing model AICs it was determined that only site was required as the random effect for all flow periods. Velocity was included as a fixed effect because in previous years it was identified as an important driver of fish food taxa. The 95% confidence intervals for the fixed coefficient of year were calculated and plotted using the R package jtools version 2.0.1 (Long 2019).

13.5 Results

Dipterans and Trichoptera made up >50% of the benthic invertebrate community in the fish stomach contents of RBT and MWF (Figure A45). Hydropsychidae (net-spinning caddisflies, Trichoptera) were a dominant family in all fish stomach samples of RBT and MWF from 2012 and 2014. Simuliidae had higher relative abundances in MWF compared to RBT. In MWF fish stomachs, Simuliidae had higher abundances in the 2012 samples compared to the 2014. Other dominant families in the MWF and RBT fish stomachs included Trichoptera and Arachnida.

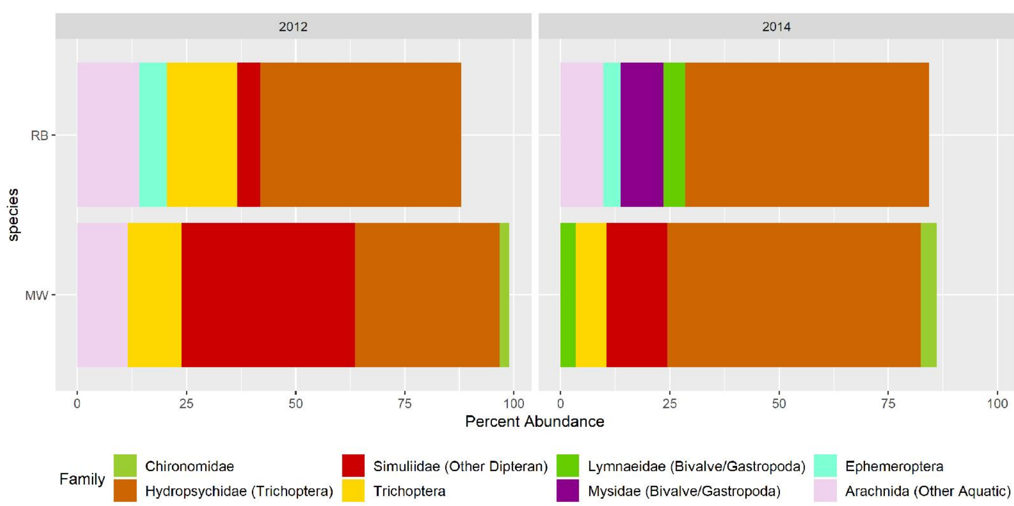

Figure A45. Dominant invertebrate taxa families for fish stomachs of MWF and RBT caught in Fall 2012 and 2014.

The dominant benthic invertebrate families from the LCR productivity sites were compared between seasons for 2012 and 2014. The winters of 2013 and 2014 had different proportions of dominant benthic invertebrate families compared to fall and summer. At the LCR sites in Winter 2013 and 2014, Simuliidae and Chironomidae were the dominant invertebrate families. In fall and summer of 2012 and 2014 the most dominant benthic invertebrate families were Hydropsychidae and Chironomidae (Figure A46). Ephemerellidae was also a dominant family at most LCR sites but had lower abundances compared to Hydropsychidae and Chironomidae. Simuliidae was a dominant family at S1 and S7 in fall of 2012.

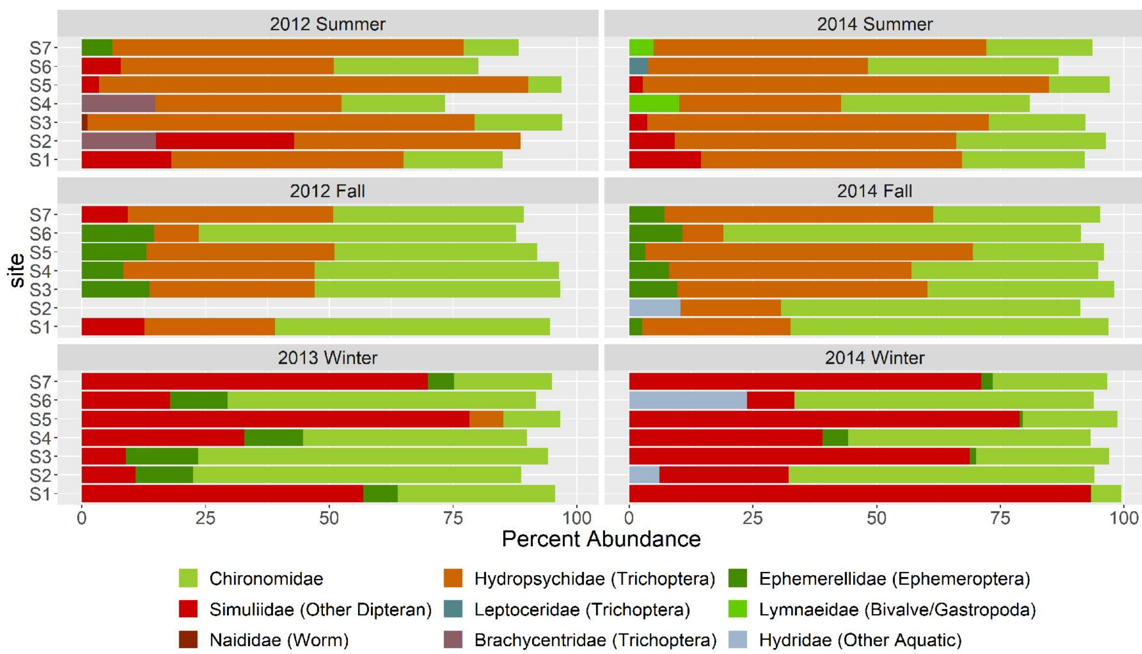

Figure A46. Dominant invertebrate taxa families of LCR benthic invertebrate samples for Summer, Fall and Winter in 2012 – 2014.

An NMDS was run on the RBT and MWF stomach contents at the family level. The stomach contents of RBT and MWF had similar benthic invertebrate compositions, with small differences between MWF and RBT fish stomachs (R^2=0.02, F=2.52, p= 0.007) and year (R^2=0.02, F=2.33, p= 0.006), as determined by a PERMANOVA. There were a few fish caught in 2014 that had distinct community compositions (Figure A47).

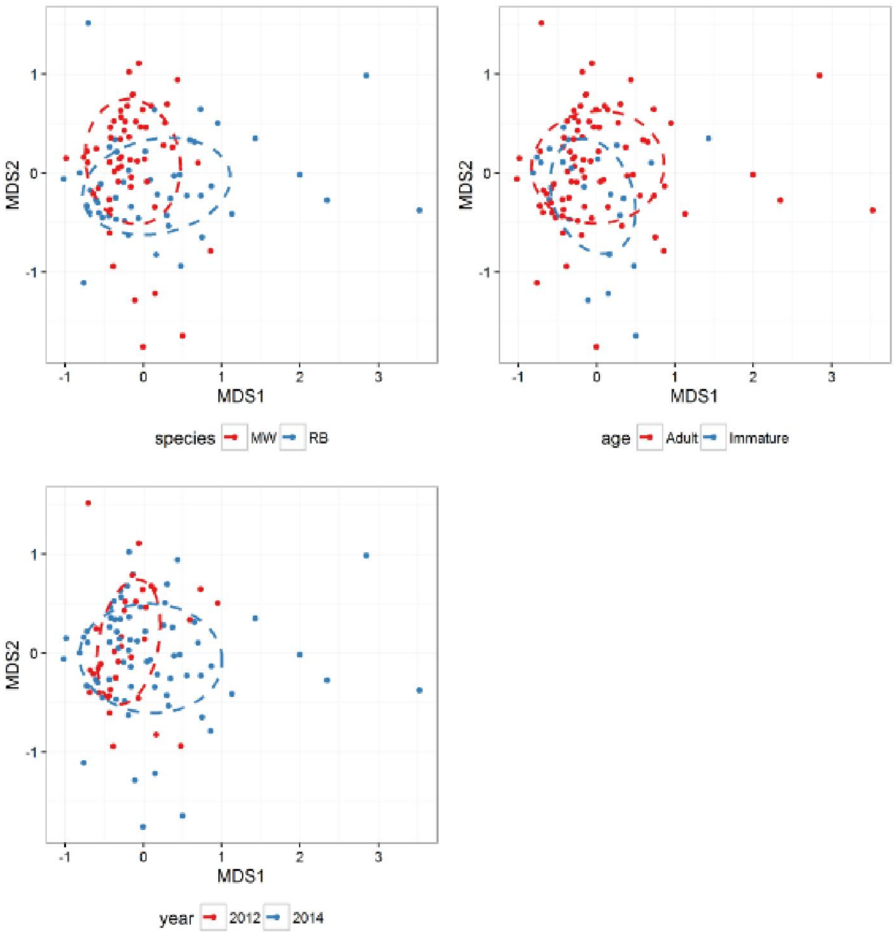

Figure A47. NMDS of benthic invertebrate community found in fish stomachs of MWF and RBT caught in Fall 2012 and 2014 grouped by species, age, and year.

The total biomass of EPT+D was used as a metric to assess the availability of fish food organisms. In linear mixed effects models this metric had a positive association with velocity during all flow periods (Figure A48). Total biomass of EPT+D also has high annual variability in all flow periods. During the MWF period, the total biomass of EPT+D was significantly higher in 2014 compared to 2013. Also, 2016 and 2018 had significantly lower total biomass of EPT+D compared to 2013 in the MWF period. During the RBT period, 2016 was the only year that had a significantly different total biomass of EPT+D compared to 2012. The total biomass of EPT+D was significantly higher in 2014 and 2016 compared to 2012 during the FFF period.

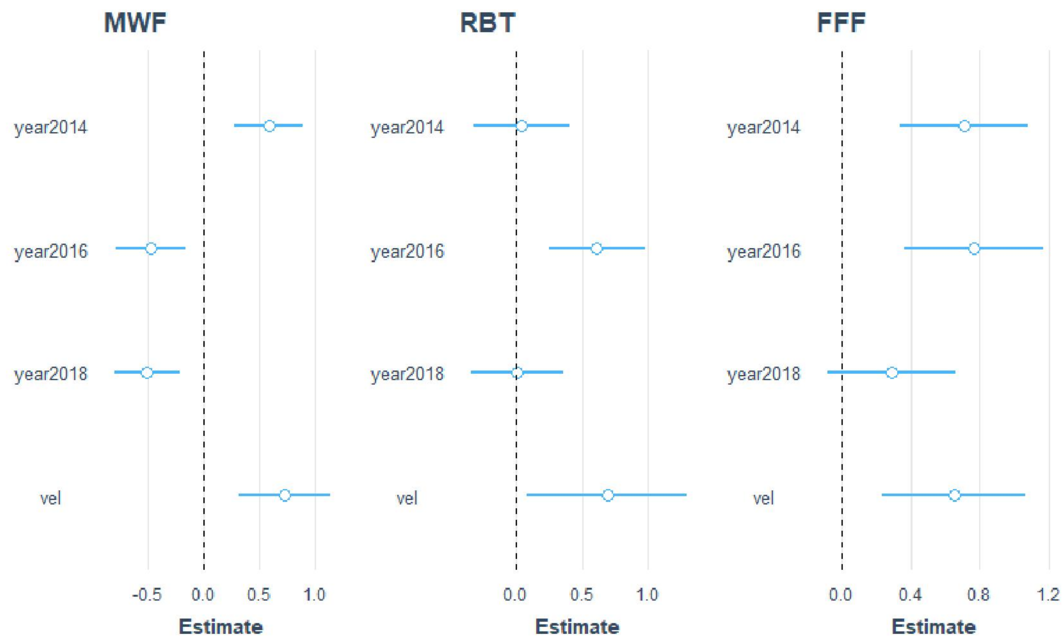

Figure A48. Total Biomass of EPT+D for MWF, RBT and FFF flow periods, fixed effects confidence intervals for velocity and year compared to reference year for MWF=2013, RBT and FFF=2012.

13.6 Discussion

The fish stomach content analysis confirmed that RBT and MWF in LCR consumed primarily Trichoptera and Dipteran in the fall, when the fish were sampled. The dominant taxa for most fish stomachs was Hydropsychidae (net-spinning caddisflies, Trichoptera) and corresponded to dominant invertebrate taxa obtained through artificial substrate sampling during the fall sampling period. The percent relative abundance of Trichoptera in juvenile and adult MWF averaged 98±4.0% and 86±31%, whereas in juvenile and adult RBT the mean percent relative of abundance of Trichoptera averaged 64±37% and 64±35%.

The analysis also showed little variation in the invertebrate taxa consumed by the two species of fish, either by age structure (juvenile vs. adult) or among years sampled. These findings are consistent with others (Vinson and Budy 2011). A diet analysis by Vinson and Budy (2011) showed that diet overlap between rainbow trout and mountain whitefish was high, and they also observed little variation in diet among years.

This exercise confirmed that the invertebrates consumed by MWF and RBT in LCR are also the dominant organisms driving the benthic productivity. Although there is no fish stomach data in the winter and summer, the dominant fall taxa were consistent with the families that were consumed in the fall. Because there is overlap with the taxa that the fish are consuming and the dominant invertebrate families sampled in the summer and winter, we can assume that MWF and RBT are mainly consuming Hydropsychidae in the summer and Simuliidae during the winter, when they are most readily available.

Velocity was also identified as an important factor in explaining variability of fish food (biomass of EPT+D). The high biomass of EPT+D in summer of 2016 can be explained by higher deployment velocities compared to other summers because of an early freshet in 2016 (Plewes et al. 2017). In other studies, the abundance of EPT taxa was also positively correlated with water velocity (Pastuchova et al. 2008).

Because the managed MWF, RBT and FFF periods cause changes to the area of submerged habitat, as well as changes to river velocities, ***we accept the management hypothesis HO_{3Aeco}, that MWF flows does not increase the availability of fish food organisms but reject HO_{3Beco} and HO_{3Ceco} that the continued implementation of RBT and FFF does not increase the availability of fish food organisms in LCR.*** During MWF managed flows, the discharge from HLK has been stabilized with much lower discharges during the first half of the flow period. These lower flows result in less wetted habitat and reductions in velocity, both of which are likely to decrease the production of fish food. In contrast, the flows during RBT and FFF are stabilized and these flows maintain a greater area of wetted habitat and higher velocities, both of which are expected to increase the availability of fish food organisms. Please refer to Appendix 7 for a more detailed discussion of the individual flow periods and their effects on the benthic invertebrate community.

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