

Columbia River Project Water Use Plan

LOWER COLUMBIA RIVER FISH

Reference: CLBMON-44

***Lower Columbia River Physical Habitat and Ecological Productivity
Monitoring – Addendum***

Study Period: 2022 - 2023

**Ecoscape Environmental Consultants Ltd.
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**Lower Columbia River Fish
Monitoring Program No. CLBMON-44
Lower Columbia River Physical Habitat and Ecological Productivity – 2022-
2023**



2023 Report

Prepared for



**BC Hydro Generation
Water Licence Requirements**

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December 2023



Suggested Citation:

Cid-Yanez, F., Kelly, A., H. Larratt, K. Hawes and M.A. Olson-Russello. (2023) CLBMON-44 Lower Columbia River Physical Habitat and Ecological Productivity Monitoring 2023. Unpublished report by Ecoscape Environmental Consultants Ltd. for BC Hydro Generations, Water License Requirements, Burnaby, B.C. 43 pp + Appendices.

Cover photos:

From left to right: 2017 Genelle; 2016 winter deployment (S7); 2019 bathymetry survey; 2022 spring data collection.

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ACKNOWLEDGEMENTS

Ecoscape would like to express our appreciation of Teri Neighbour, Margo Sadler, and Dr. Guy Martel of BC Hydro, who have provided administrative guidance and scientific oversight.

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EXECUTIVE SUMMARY

The goal of this report is to assess the level of certainty in estimating the effects of Rainbow Trout Spawning Protection Flows (RBTSPF) on primary productivity. This work is an addendum to CLBMON-44, and it focuses on modelling and measuring benthic productivity at Norns Creek (NC) fan, between April 1st and June 30th, for years 2022 and 2023. Benthic productivity was modelled using a 2D hydraulic model and survival curves previously developed. Measurements of benthic productivity were carried out as previous years, with artificial samplers deployed at three new sites throughout NC fan, and one control site (R2-S1) in the channel along the opposite bank from NC fan. Water stage (elevation), temperature, light intensity, turbidity, substrate size, and velocity were also measured. Both years, 2022 and 2023, were managed as OFF years (no RBTSPF).

More stable flows in the second half of the RBTSPF period in 2022 allowed for higher estimated productivity than in 2023. Regardless, flow variability reflected the absence of RBTSPF in 2022 and 2023. Overall, accrual of invertebrate dry biomass was higher during ON years (254 ± 48.5 Kg, $n = 7$) than during OFF years (201 ± 47.7 Kg, $n = 8$) (p -value = 0.02, effect size = 0.98). Modelled accrual of periphyton (chlorophyll-a, chl-a) for ON (18.5 ± 3.5 Kg) and OFF years (15.8 ± 3.91 Kg) did not significantly differ (p -value = 0.09, effect size = 0.7). Uncertainty on the effectiveness of RBTSPF, as captured by the modelled results and expressed in terms of statistical power, decreased with the added data. Power of the test on the difference of means, went from 0.51 to 0.7 for invertebrate biomass and from 0.4 to 0.5 for chl-a.

The invertebrate community in the Lower Columbia River (LCR) is typical of regulated river systems. NC fan is dominated by taxa that are well adapted to varial zone habitats and can resist desiccation when flows drop. These taxa include Chironomidae and Naididae, and certain Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa. Observed benthic community results were influenced by specific events affecting each year. Wildfires affected the immediate watershed in 2021 and local landslides occurred in Norns Creek fan in 2023. Although, variability was comparable to previous observations in the LCR, benthic communities in the NC fan differed from other LCR sites. Functional feeding groups, distributions of dominant taxa, and taxa richness in NC sites differed from the control and previously surveyed sites. These differences are likely driven by habitat types, particularly substrate size. NC sites have small substrates with less interstitial space, whereas R2-S1 (control site) substrates are larger cobble and boulder, reflecting differences in velocity and flow patterns.

Periphyton productivity in the NC fan exceeded previous LCR measurements in 2022, during a *Didymo/Ulothrix* bloom, but was similar to the LCR control site in 2023, despite scouring from a bank failure in the fan. Regardless, periphyton data reflect that Norns Creek fan is a productive and diverse site within the LCR. *Didymo* productivity metrics did not differ significantly between ON and OFF years. Periphyton growth in the Norns Creek fan was driven more by the *Didymo* bloom and a 2023 erosion event than by RBTSPF periods in the 2022-2023 study period.

The measured productivity of samples collected during OFF (2 in spring) and ON years (8 in multiple seasons) provide a reference for modelled data but do not allow for a comparison with modelled results (spring). NC fan RBTSPF benthic productivity measurements were only available for 2022 and 2023, and results were marked by the *Didymo* bloom and land slides, respectively.

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1 INTRODUCTION

Discharges from the Hugh L. Keenleyside Dam (HLK) and Arrow Lakes Generating Station (ALH) during winter and spring have the potential to affect salmonid spawning and rearing habitats. To minimize impacts, BC Hydro may alter operations of HLK and ALH to create Rainbow Trout (*Oncorhynchus mykiss*) spawning protection flows (RBTSPF) which stabilize or increase discharges from April 1 through June 30, and thus reduce redd dewatering and subsequent Rainbow Trout egg losses as well as maintain rearing habitat (BC Hydro 2007).

An initial twelve-year study of the physical habitat and ecological productivity on the Lower Columbia River (LCR) (CLBMON-44), between the combined outflow of the HLK Dam Spill (HLK) and Arrow Lakes Generating Station (ALH) and the Birchbank gauging station (BBK) near the southern British Columbia border was finalized in 2019. A final summary report of hydrological and benthic productivity data collected between 2008 and 2019 was issued to BC Hydro and Power Authority (BC Hydro) in August 2019 (Olson-Russello et al. 2019). One objective of CLBMON-44 was to examine the influence of the RBTSPF periods on physical habitat components and benthic productivity measures, such as benthic invertebrates and periphyton biomass, as primary food sources for fish.

Despite the implementation of RBTSPF, it was unclear how these flows affected the local RBT population abundance (BC Hydro, 2019). To reduce uncertainty, an experimental approach of alternating RBTSPF with standard operating flows in alternating years (e.g., ON and OFF RBTSPF) was initiated in 2019 for a maximum duration of five years, until 2023. In this initial extension, the focus was to assess how the RBTSPF (ON), or lack thereof (OFF) may affect the ecological productivity of Norns Creek (NC) fan, an important spawning habitat for RBT. Ecological productivity was represented by benthic productivity. Benthic biomass, inclusive of periphyton and macroinvertebrates, is key in river systems because these organisms are a primary food source for fish, specially in early developmental stages.

The impact of RBTSPF on benthic productivity was assessed using a hydraulic model developed using Telemac-2D (Open Telemac-Mascaret Consortium 2020) and accompanying productivity curves (Plewes et al. 2020). The productivity models used hourly depths from the hydraulic model and growth/colonization and death curves for invertebrates and periphyton. Contingent with modelling productivity, in-situ measurements focused on the NC fan were carried out.

This 2-year extension contract builds on the productivity modelling undertaken in 2019 and 2020 adding benthic sampling in the Norns Creek fan. To further understand the impact of RBTSPF and increase certainty, the following work plan was developed:

- Update the existing model of ecological productivity for 2022 and 2023, with a focus on benthic invertebrate species that are most likely to be consumed by juvenile Rainbow Trout.
- Monitor periphyton and benthic invertebrate communities within the Norns Creek fan with the use of artificial benthic samplers deployed during the RBT flow period.
- Continue the maintenance and collection of water level stage data for all pre-established sampling locations in 2022 and 2023, and
- Continue monitoring photosynthetic active radiation (PAR) and turbidity.

1.1 Literature Review

The growth and mortality curves used in the productivity model were adapted from curves developed for the Columbia Power Corporation (CPC) (Schleppe et al., 2013). These CPC models were derived from benthic data collected in the LCR and considered other mortality curves from the literature (Schleppe et al., 2013). These models used invertebrate dry weight and chl-a concentrations for periphyton by sampling these parameters at 6, 12, 24, 48 and 64 days of incubation on the LCR and Kootenay rivers (Schleppe et al., 2013). Several curves were used to produce one growth and one mortality curve for periphyton, and one growth and one mortality curve for benthic invertebrates. The invertebrate taxa used to create these mortality curves were Simuliidae (blackfly) and Hydropsychidae (net-spinning caddisfly), two of the most abundant LCR invertebrate taxa (Schleppe et al. 2013, Schleppe et al. 2015).

The previous works by BC Hydro and the CPC have also considered preferred forage items for RBT. During the last 12 years of benthic invertebrate sampling in LCR, *Chironomidae* (non-biting midges), *Hydropsychidae*, and *Simuliidae* together make up more than 86, 80 and 79 percent of invertebrates present in the LCR in winter, summer, and fall (Olson-Russello et al. 2019). In general, *Diptera*, *Ephemeroptera* (mayflies), *Plecoptera* (stoneflies), and *Trichoptera* (caddisflies) make up the largest proportion of fish food, for all life stages.

The growth/mortality curves used in the model are specific to LCR taxa that constitute a large portion of RBT diets; *Simuliidae* and *Hydropsychidae*. Furthermore, the mortality curves were derived from data collected in the LCR, so they reflect the specific site conditions of the river system. After reviewing the literature, we concluded that the effort and associated cost of producing and integrating mortality curves for more taxa would not significantly increase model precision. The literature review and more detailed conclusions can be found in Appendix A.

2 STUDY AREA

The CLBMON-44 study area is in southeast British Columbia on LCR between HLK Dam and the BBK gauging station. Kootenay River is a major tributary to the 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. The five Water Quality Index Stations (WQIR2-S1-5) on LCR are located between the HLK Dam and BBK gauging station (Figure 2-1).

The focus of this two-year study is the Norns Creek confluence at the downstream end of Reach 1 (Figure 2-2). Water quality index station 2 (WQIS2) is located approximately 1 km upstream of the Norns Creek confluence, and WQIS3 is about 800 m downstream of the confluence. The hydraulic model extent includes the whole river between WQIS2 and 3. The productivity model extent includes the Norns Creek fan where Rainbow Trout redds have been documented.

Productivity sampling in 2022 and 2023 included three sites (NC-1 through NC-3) with 5 samplers per site (15 samplers) in the Norns Creek fan and one control site (R2-S1) with 5 samplers in LCR, in the channel along the opposite bank from Norns Creek fan (Figure 2-2). R2-S1 is an appropriate control site because it is the closest to the Norns Creek Fan, and it is approximately the same distance from HLK Dam. Although both sites experience similar flow fluctuations from the Dam, Norns Creek produces a different local hydrodynamics in the fan. Consequently, substrates are different between the two sites, and R2-S1 is not RBT spawning habitat. The difference in substrates between Norns Creek fan and the control R2-S1 is important to consider when comparing benthic community metrics, as substrates and flows drive the benthic invertebrate community and productivity.

The Norns Creek fan is a morphologically different habitat than R2-S1 and other downstream habitats sampled in previous years. The Norns Creek site is a large alluvial fan at the confluence with the LCR. Substrates in Norns Creek fan are pebble, gravel, and sand. Conversely, the R2-S1 site is in a more confined section of the river with a steep erosional bank and substrates are dominated by cobble and boulder. Hydraulic lift at the upstream end of the fan deflects more intense current to the right bank with a more smooth and laminar flow occurring as the LCR spills over the finer textured fan. The Norns Creek fan represents the natural control outlet from the Robson Reach of the LCR, which is lacustrine in character. The channel is then constricted with a slight increase in gradient downstream along the R2-S1 site as the river spills from the Robson Reach downstream.

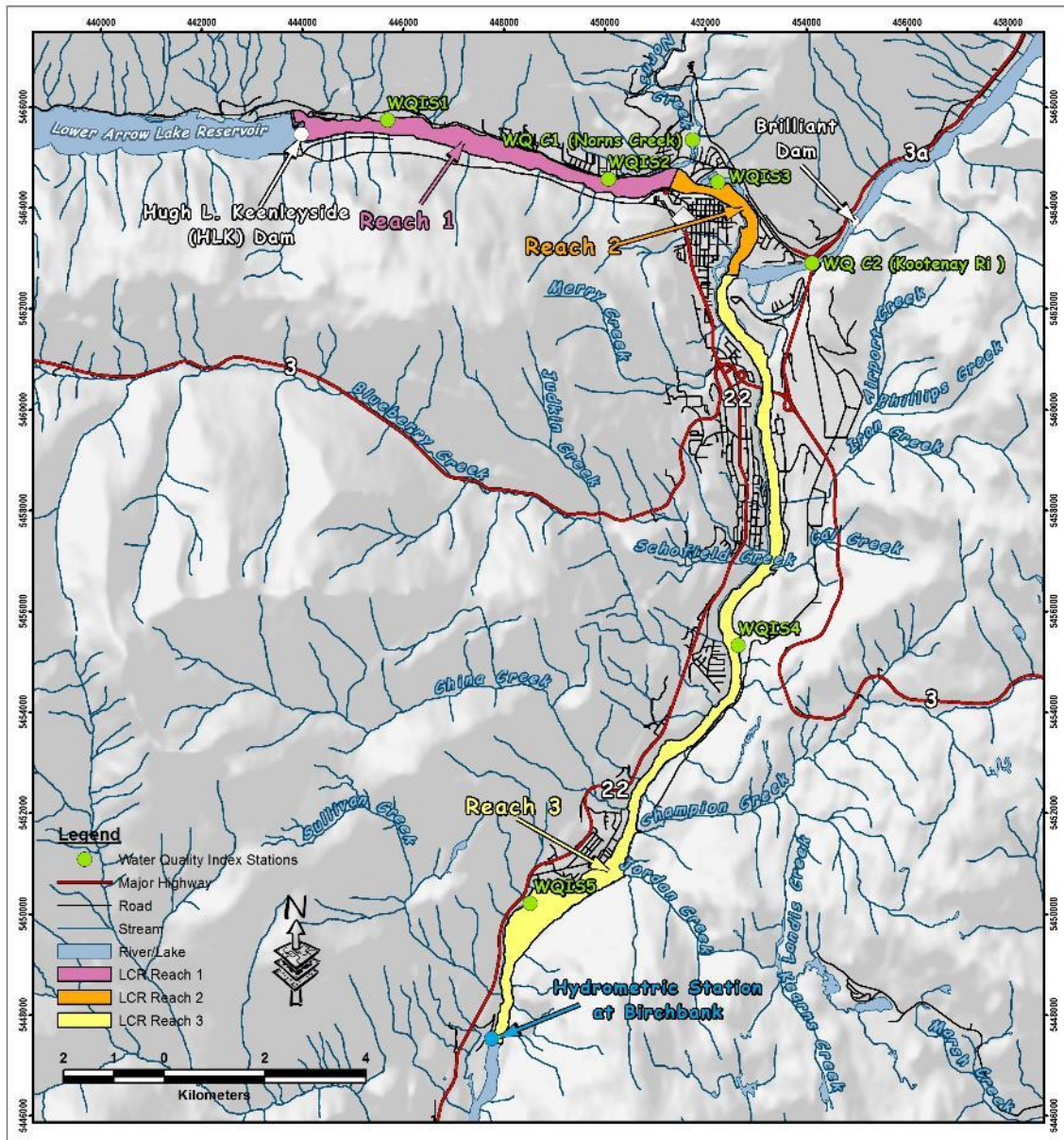


Figure 2-1: The Lower Columbia River (LCR) study area.

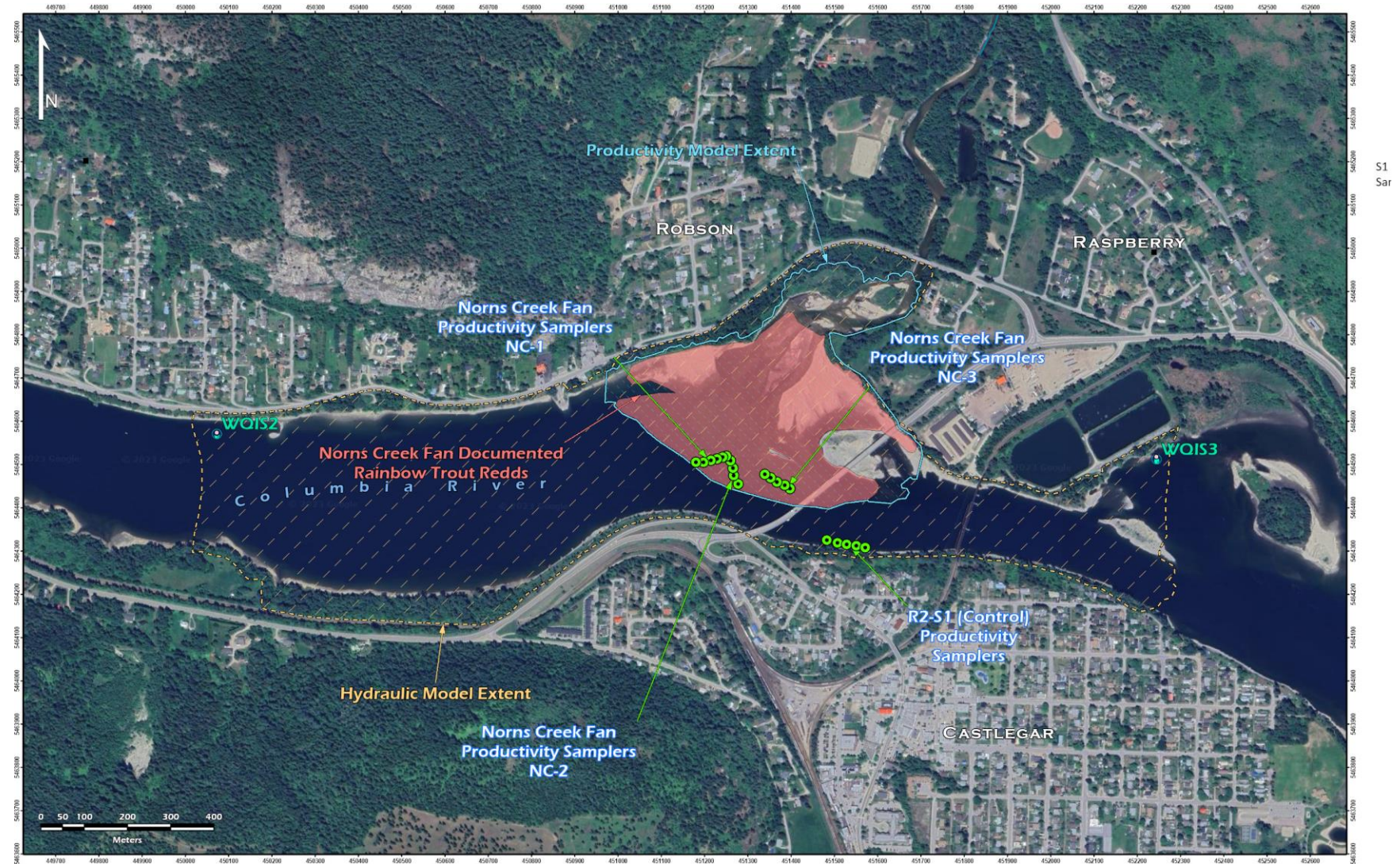


Figure 2-2: The Norns Creek fan study area and R2-S1 sample site.

3 METHODS

The project’s methods include two paths. First, discharge data was collected and used to simulate hydrological dynamics and estimate productivity (accrual) of benthic invertebrates and periphyton biomass in the study area during RBTSPF periods. Second, artificial substrate devices were deployed to empirically measure the accrual of periphyton and benthic invertebrates and to assess environmental conditions at different sites in the study and control areas.

3.1 Hydrological and Physical Characterization

3.1.1 Discharge Data and Flow Metrics

Time series data for combined discharge of HLK Dam spill and Arrow Lakes Generating Station (HLK_ALH), and Birchbank WSC station were collected and summarized. Flow metrics, including coefficient of variation (cv), maximum, minimum, median and mean daily flows, were computed for the period of interest, and compared to previous years. Hydrographs of the combined discharge (HLK_ALH) mean daily flows from April 1st to June 30th (RBTSPF) were also generated for 2022 and 2023 and compared with previously modelled ON and OFF years. To better understand the flow dynamics leading to the RBTSPF period, flow years are started on both, January (calendar year), and July (water year).

3.1.2 Water Level Data Download and Maintenance

Level loggers data collection was restarted at LCR water quality index stations (WQIS1-5) and at a single station on the Kootenay River (WQC2) (Table 3-1, Figure 2-2) before the onset of each RBTSPF period. Level loggers recorded water elevation and water temperature in hourly intervals. Two new level loggers were purchased to replace aging loggers from the original CLBMON-44 program (Olson-Russello et al. 2019). The new loggers were installed at WQIS2 and WQIS3, as these stations are located at the upstream and downstream end of the hydraulic model extent. Elevation data from WQIS3 was used in the calibration of the hydraulic model. Water level data for both modelling years was downloaded during the RBTSPF and at the end of the period. Level loggers were left in place for data to be continually recorded and downloaded again. Water elevation was determined using Hoboware sensors (Onset Computer Corporation, 2023) with reference barometric data from two stations placed in proximity of the level loggers.

Table 3-1: Water level monitoring station locations.

Station Name	Location Description	Station Characteristics	UTM Coordinates	
			Northing	Easting
WQIS1	Across from Zellstoff Celgar Ltd.	Upstream of Celgar outfall	5,465,742	445,693
WQIS2	Upstream of boat launch	Downstream of Celgar outfall	5,464,573	450,072
WQIS3	Downstream of railway bridge	Within back-channel area	5,464,517	452,244
WQIS4	~7 km downstream of Kootenay River confluence	Left bank on bedrock face	5,455,332	452,653
WQIS5	~ 2.2 km upstream of Birchbank	Right bank on bedrock face	5,450,221	448,514
WQ C2	Kootenay River, just above confluence with LCR	Right bank, on bedrock face	5,462,911	454,114

3.1.3 PAR, Turbidity Profiles, and Light Analysis

Photosynthetic active radiation (PAR), or light intensity, was measured at WQIS2, WQIS3, Robson boat launch, and shallow and deep sites in the Norns Creek fan in April and June of both experimental years. Measurements were taken at the surface and at 50 cm intervals throughout the water column using a PAR metre. In situ turbidity was simultaneously measured at each interval to determine the continuous light attenuation coefficient based on turbidity. The profiles extended to depths that ranged from 2 to 10 m at the five sites.

The light intensity, PAR, and turbidity data for the five sites were used to model light attenuation and estimate the depth of the photic zone for the Norns Creek fan study area. The euphotic depth is defined as the depth at which light in the photosynthetic active range (PAR) is attenuated to 1% of its surface value (Lee et al. 2007). Light availability model parameters were estimated using Bayesian methods with JAGS (Plummer 2003). Unless otherwise indicated, the Bayesian analyses used weakly informative normal and half-normal distributions (Gelman et al., 2017). The posterior distributions were estimated from 1500 Markov Chain Monte Carlo (MCMC) samples thinned from the second halves of three chains (Kery and Schaub 2011, 38–40). Model convergence was confirmed by ensuring that the potential scale reduction factor $\hat{R} \leq 1.05$ (Kery and Schaub 2011) and the effective sample size (Brooks and Johanson 2011) $ESS \geq 150$ for each of the monitored parameters (Kery and Schaub 2011).

The parameters are summarised in terms of the point estimate, lower and upper 95% compatibility limits (Rafi and Greenland, 2020) and the surprisal s-value (Greenland 2019). The estimate is the median (50th percentile) of the MCMC samples while the 95 % CLs are the 2.5th and 97.5th percentiles. The s-value indicates how surprising it would be to discover that the true value of the parameter is in the opposite direction to the estimate (Greenland 2019). An s-value > 4.3 bits, which is equivalent to a significant p-value < 0.05 (Kery and Schaub 2011; Greenland and Poole 2013), indicates that the surprise would be equivalent to throwing at least 4.3 heads in a row.

The condition that parameters describing the effects of secondary (nuisance) explanatory variable(s) have significant p-values was used as a model selection heuristic (Kery and Schaub 2011). Based on a similar argument, the condition that random effects have a standard deviation with a lower 95 % compatibility interval (CL) > 5 % of the estimate was used as an additional model selection heuristic. Model adequacy was assessed via posterior predictive checks (Kery and Schaub 2011). The analyses were implemented using R version 4.2.2 (R Core Team 2022) and the `mbx` family of packages (Poisson Consulting Ltd, 2023). Further details of the analysis are provided in Appendix B.

3.2 Hydraulic and Benthic Productivity Modelling

3.2.1 Hydraulic Modelling

A TELEMAC-2D hydraulic model created in 2019 (Plewes et al. 2020) was used to simulate water depth and velocity given measured discharge volumes and reference elevations recorded for the corresponding years. Model simulations used hourly discharge data from HLK_ALH (combining HLK Dam Spill, and Generating Station, ALH) and hourly elevations from the WQIS3 level logger for the RBTSPF period of the corresponding year. A detailed methodology of the hydraulic model development and analysis of previous years data is available in Plewes et al. 2020. Additional flow sources such as Norns Creek or backwater from downstream of the fan, are not included in the model. Model configuration files, including bathymetry and boundary conditions, are provided in the data package. Backwatering was not directly quantified but indirectly considered by including the water elevation at the exit of the modelled study area. All simulation inputs and results are stored in a relational database.

3.2.2 Productivity Modelling

The biomass productivity models use asymptotic growth curves with dewatering mortality and carrying capacity components for periphyton and benthic invertebrates derived from previous desiccation survival experiments and Columbia Power Corporation productivity studies (Schleppe et al. 2013; 2015). The invertebrate mortality curve was based on data from desiccation of *Chironomidae*, *Simuliidae* (blackflies), and *Hydropsychidae* (net-spinning caddisfly), three of the most abundant taxa in the LCR (Schleppe et al. 2013, Olson-Russello et al. 2019). Invertebrate dry biomass and chl-a in 2022 and 2023 were predicted using hourly depth results from the hydraulic model to estimate submerged/exposed times. Submerged time increased productivity of a given area, while exposed time increased periphyton and invertebrate mortality. The invertebrate dry biomass and periphyton chl-a modeling for both years started on April 1st at 0:00 at the minimum values of 1 mg/m² and 0.05 µg/cm², respectively. Total chl-a and invertebrate biomass accrual in kg were estimated at the end of RBT flow period on June 30th at noon.

Modelled invertebrate and chl-a productivity were then compared between ON and OFF RBTSPF years. A one-sided two-sample t-test with a 90% C.I. was used to determine if chl-a and invertebrate biomass accruals were higher during ON years compared to OFF years. A power analysis was also conducted to determine the statistical power of the t-test using the *pwr* package in R (Champely et al. 2020). Due to the small sample size ($n_{\text{OFF}} = 8$, $n_{\text{ON}} = 7$) a significance level of 0.1 was used to increase the statistical power of the test. Shapiro-Wilk and Levene's tests were performed to confirm the assumptions of normality and equal variance of sample groups.

3.3 Benthic Productivity Sampling

In the last two study years, 2022 and 2023, the CLBMON-44 program collected benthic productivity data directly within the Norns Creek fan. Previously all productivity data was collected in the mainstem of the LCR, mostly within Reach 2. These were also the first years that productivity samplers were deployed to overlap the RBTSPF period (spring). Samplers were in the river from Apr 1 – Jun 30 in 2022, and from April 5 – June 28 in 2023. Historically, summer samplers were deployed near June 1 and were retrieved in mid-August. Although the focus of this study was the Norns Creek fan, a previously sampled site in Reach 2 (R2-S1), was established as

productivity control site. The R2-S1 control provides the context to interpret the inherent seasonal and annual variability of periphyton and invertebrate data.

The objective of the NC fan productivity sampling was to understand how the benthic community may or may not differ from other previously sampled sites on LCR. This was deemed important because the model employed to estimate the effects of ON or OFF RBTSPF on the NC fan productivity used growth and death curves derived from periphyton and invertebrate data from other LCR sites. Benthic productivity is largely driven by physical factors such as substrate size, velocity, light, and water depth. Norns Creek fan differs from most of the other LCR sites sampled previously in that it has a low gradient, smaller substrates, less velocity, and a more laminar flow. Therefore, this study of Norns Creek fan sites provides additional data to determine to what degree the benthic communities of the fan differed from the other LCR sites.

3.3.1 Data Collection

Benthic productivity of the Norns Creek fan was investigated using artificial Styrofoam samplers for periphyton and rock baskets for benthic invertebrates. A typical design of the periphyton and macroinvertebrate sampling apparatus is illustrated in Figure 3-1. The three Norns fan sites differed from the typical design in that a single anchor and buoy were used for all five samplers. At the time of sampler deployment and retrieval, velocity and depth at each sampler were also recorded. Sampler placement was determined based on flows and velocities and where RBT spawning sites were identified in the field. In 2022, placement avoided exposed portions of the fan and areas where water velocities were very low or backwatered and thus not conducive to RBT spawning. However, in 2023, placement was adapted to include exposed sites in early spring but known to be submerged later in the season.

Three sets of five samplers were deployed in the NC fan, and one set of five samplers was deployed at site R2-S1 (Figure 2-2). Samplers were placed at increasing wetted depths from approximately 0 to 4.5 m (Table 3-2). The samplers were left in place for 13 weeks to capture the entire RBT flow period. Depths at retrieval, ranged from 2 – 6.9 m (Table 3-2).

At the end of each RBTSPF period, three periphyton Styrofoam punches were randomly collected from each sampler plate to assess 1) chl-a; and 2) taxonomy 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 (Olson-Russello et al. 2019). In 2022 the recovery rate was 100% for all samplers (Table 3-3). In 2023, the rock basket from the moderately deep (MD) sampler of NC-2 broke open upon retrieval and no benthic invertebrate sample was retrieved from it.

Table 3-2: Naming convention of sampling depths and corresponding depth strata for 2022-2023 samples.

Depth Label	Depth Name	Depth at Deployment (m)	Depth at Retrieval (m)
D	Deep	2.0 – 4.5	3.8 – 6.9
MD	Moderately deep	1.6 – 2.2	3.5 – 5.5
M	Mid	1.5 – 1.8	2.8 – 3.4
MS	Moderately shallow	1.1 – 1.5	2.7 – 2.8
S	Shallow	0 – 0.8	2 – 2.2

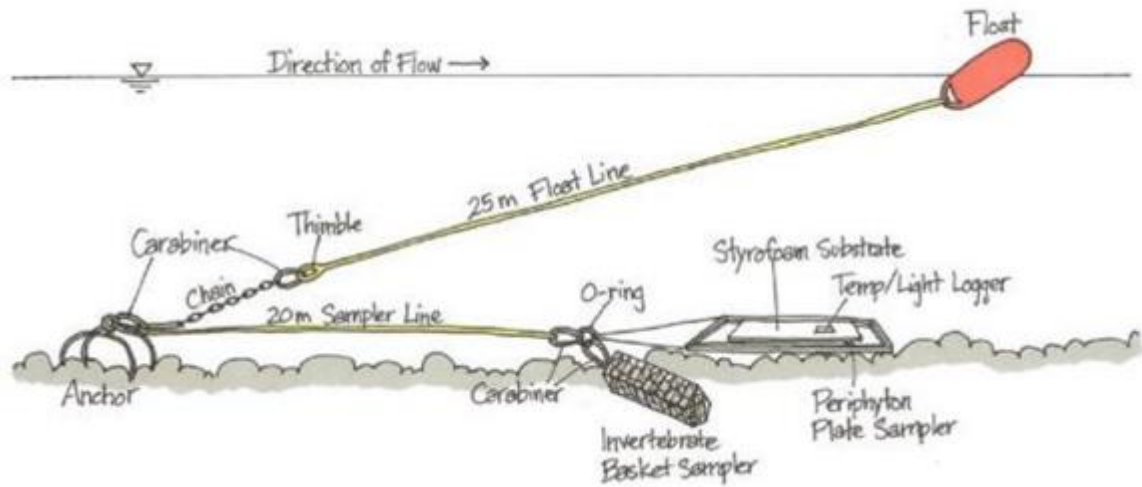


Figure 3-1: Typical benthic productivity sampler.

Table 3-3: Artificial sampler deployment and recovery rates in 2023.

Season	Reach	Site	Periphyton Samplers		Invertebrate Basket Samplers	
			# Deployed	# Retrieved (%)	# Deployed	# Retrieved (%)
RBT Flow Period (Apr 5-Jun 28) 13 weeks	1 & 2	Site 1 (R2-S1)	5	5 (100)	5	5 (100)
		Site 2 (NC-1)	5	5 (100)	5	5 (100)
		Site 3 (NC-2)	5	5 (100)	5	4 (80)*
		Site 4 (NC-3)	5	5 (100)	5	5 (100)
Totals			20	20 (100)	20	20 (95)

* The NC-2 MD rock basket broke open and was lost during retrieval.

3.3.2 Laboratory Taxonomic Identification and Processing

Of the three 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. The remaining two punches were used for taxonomic identification by Larratt Aquatic (LAC) in West Kelowna. Fresh, chilled punches were examined within 48-hours for protozoa and other microflora that cannot be reliably identified from preserved samples. One of the two punches was frozen and stored until taxonomic identification and biovolume measurements could be undertaken. Species cell density and total biovolume were recorded for each sample. A photographic archive was compiled from LCR samples. Detailed protocols on periphyton laboratory processing are available from LAC. Periphyton datasets were standardized for subsequent analyses.

Following retrieval, fixed benthic invertebrate samples were transported to Cordillera Consulting (CC) in Summerland BC. Samples were sorted and identified to the genus-species level where possible following standard procedures. 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 SAFIT (Richards & Rogers, 2011), CABIN (Martens et al., 2020), and others. Species abundance and biomass were determined for each sample. If samples were large, subsamples were processed following similar methods. Detailed protocols on invertebrate laboratory processing are available upon request.

3.3.3 Data Analysis

To measure and describe productivity in Norns Creek fan and at R2-S1 for 2022 and 2023, the metrics in Table 3-4 were calculated. Non-metric multidimensional scaling (NMDS) and Bray-Curtis dissimilarity were used to evaluate variations in periphyton and invertebrate community compositions across sites and sampled periods. NMDS was performed at the family level for both invertebrates and periphyton. Because each site had a transect of five samplers from shallow to deep, each sample represents a pseudo-replicate. Furthermore, the three sites in Norns Creek fan were sampled in spring of 2023 for only the second time, so the resulting dataset did not have sufficient statistical power for the previous analysis method of linear mixed effects models.

Table 3-4: Metrics for periphyton and benthic invertebrates analysed in 2023.

Variable	Description	Unit
Total Abundance	Total abundance across all species (periphyton and invertebrates)	Invertebrates: #/basket, Periphyton: cells/cm ³
Total Biovolume	Total biovolume across all periphyton species	cm ³ /m ²
Total Biomass	Total invertebrate biomass for taxa groups. For Chl-a it represents the amount in a sampled punch.	Invertebrates: g/basket Chl-a: µg/cm ²
Effective Number of Species	A measure of community diversity that is the e ^S . S= Shannon-Wiener index.	#
Species Richness	Number of unique species	#
Dominant Taxa	Dominant taxa by biovolume (periphyton), biomass (benthic invertebrates), and abundance	Taxa by cm ³ /m ² , taxa by g/basket, and #/basket

3.4 Datasets

The primary datasets collected or generated are summarized in Table 3-5.

Table 3-5: Datasets used or generated as part of the CLBMON-44 Extension Program.

Name/Description	Source	Frequency of Collection	Unit
Physical Datasets			
LCR / Kootenay River Elevation / Water Temperature	Data collected at 5 stations (LCR) and 1 station (Kootenay River)	Twice annually	°C
Hourly Discharge from HLK Dam (Spill + Generating Station, ALH)	Data obtained from BC Hydro	Continuous	m ³ /s
Light Intensity, turbidity, and depth profiles	Field data	Collected twice during RBTSPF	Photons m ² /sec, NTU, depth in m
Modelling Datasets			
Hourly depths for 1 m ² cells	TELEMAC-2D model	April- June (RBTSPF periods) of modelled years	m
Daily chl-a and invertebrate dry weight estimates by 1 m ² cells	Productivity model	April- June (RBTSPF periods) of modelled years	kg
Total chl-a and invertebrate dry weight estimates (RBTSPF accrual)	Productivity model	End of RBTSPF period (Noon of June 30 th) of modelled years	kg
Productivity Datasets			
Light / Temp	Data collected at each productivity sampler	Spring deployment and retrieval	PAR, °C
Benthic Invertebrates	Data collected at each productivity sampler. Data includes abundance, biomass, taxonomy, and metrics	Spring retrieval	# individuals/basket, g/basket, # of taxa, dominant taxa
Periphyton	Data collected at each productivity sampler. Data includes abundance, biovolume, taxonomy and chlorophyll-a	Spring retrieval	# cm ³ , cm ³ /m ² , # of taxa, dominant taxa, µg/cm ²
Velocity	Data collected at each productivity sampler twice per deployment period	Spring deployment and retrieval	m/s
Substrates	Substrate percentage at each deployment site estimated during deployment	2022	% cover

4 RESULTS AND DISCUSSION

4.1 Hydrological and Physical Characteristics

4.1.1 Discharge and Flow Variability

Discharge data was collected and analyzed for years starting in 1984 up to 2023. Additionally, statistical differences were explored for years with (ON), and without (OFF) RBTSPF, focusing on years with modelled productivity responses (1988 – 1991, 2003 - 2005, 2010, 2014, 2016, 2019, 2020, 2022, and 2023). Flows in 2022 RBTSPF period (OFF) had a mean of 746.7 ± 298 , a minimum of 285 and a maximum of $1,764 \text{ m}^3/\text{s}$. More variability was observed when extending to 2022 water year period (Figure 4-1). On the other hand, 2023 RBTSPF (also OFF) discharge statistics were relatively low compared to other OFF years, with a mean of $641.7 \pm 266.7 \text{ m}^3/\text{s}$, a minimum of 283.1, and a maximum of $1,308.4 \text{ m}^3/\text{s}$. These dynamics were also observed when considering the full 2023 water year, with flows below average preceding the RBTSPF period (Figure 4-2).

HLK daily mean flows were more variable during OFF years than in ON years during RBTSPF periods (Figure 4-3). The mean coefficient of variation for daily flows was 0.46 ± 0.13 for OFF years and 0.29 ± 0.16 for ON years. Typically, OFF years had higher daily maximum flows and lower daily minimum flows compared to ON years (Figure 4-4). The highest daily maximum flows of the OFF years were $2,544.1 \text{ m}^3/\text{s}$ in 1987, $2,009.5 \text{ m}^3/\text{s}$ in 2020 and $1,981.9 \text{ m}^3/\text{s}$ in 1988 (Table 4-1).

There were four ON years (1992, 1994, 1996, 2015) when HLK flows exceeded $1,500 \text{ m}^3/\text{s}$ during the RBTSPF period (Table 4-1) and were considered high flow years. The WQIS3 logger elevation could not be accurately predicted for years with flows over $1,500 \text{ m}^3/\text{s}$, resulting in greater uncertainty in accrual estimates for these years. Thus, high flow years were not included in the productivity modelling. The mean daily flows for years selected for the productivity model are shown in Figure 4-4.

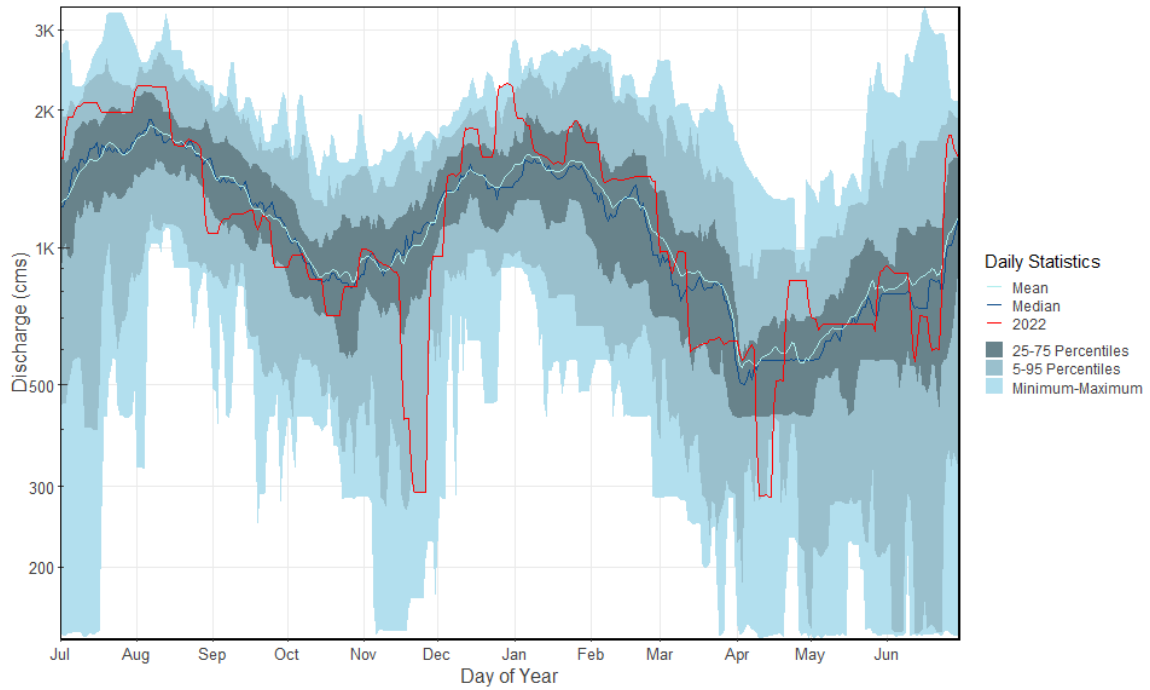


Figure 4-1: Hydrograph for HLK_ALH water year 2022 (July 2021-June 2022). OFF year

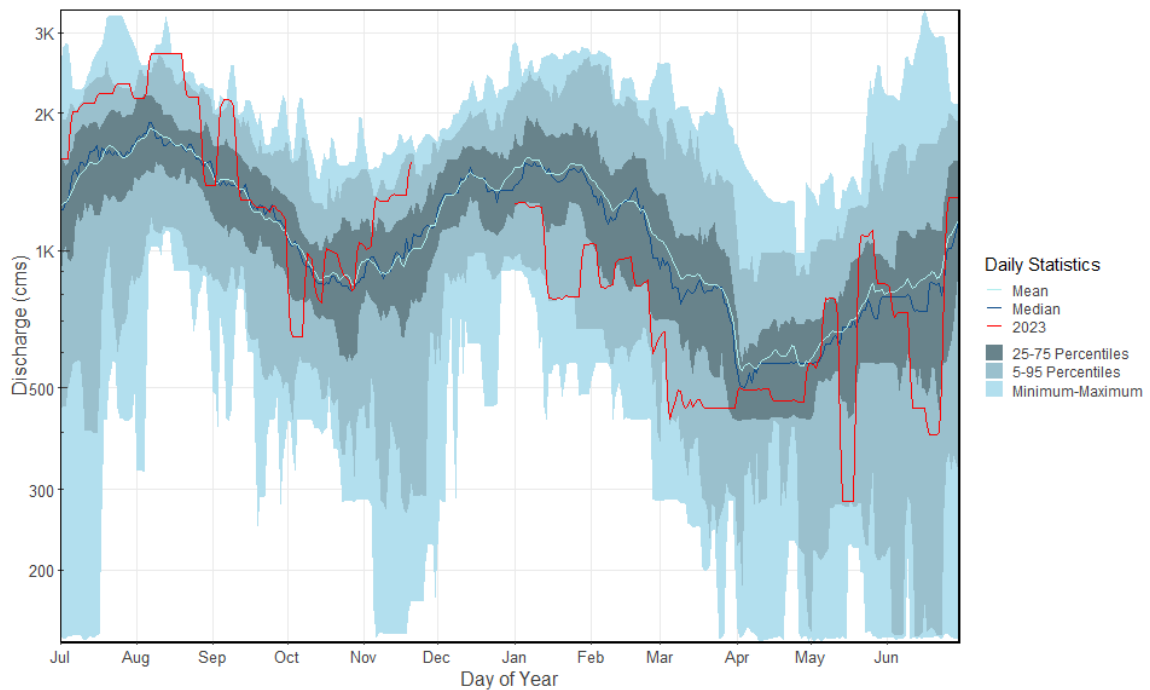


Figure 4-2: Hydrograph for HLK_ALH water year 2023 (July 2022-June 2023). OFF year

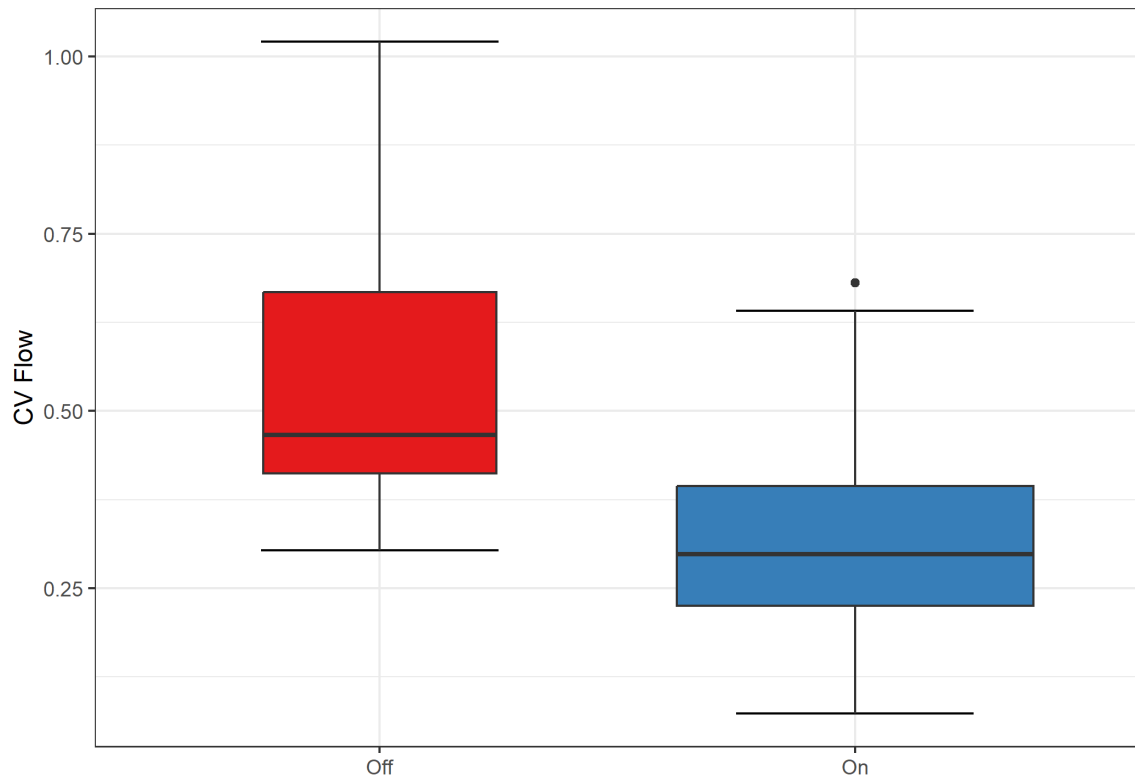


Figure 4-3: Coefficient of Variation (CV) for HLK_ALH discharge during RBTSPF OFF years (1984-1991, 2019, 2020, 2022, and 2023) and ON years (1992-2018).

Table 4-1: Flow metrics for daily mean HLK_ALH discharge flows (m³/s) during OFF and ON RBT (Apr 1 – Jun 30) flow years, including standard deviation (SD) and coefficient of variation (CV).

Year	Management	Mean	SD	Median	Min	Max	CV
1984	Off	480.9	490.7	148.7	138.6	1,480.9	1.0
1985	Off	512.6	334.7	469.5	140.7	1,277.5	0.7
1986	Off	665.8	323.5	683.2	185.8	1,429.7	0.5
1987	Off	874.2	709.9	760.3	138.7	2,544.1	0.8
1988	Off	721.6	441.3	714.0	140.5	1,981.9	0.6
1989	Off	588.7	255.5	569.0	227.7	1,094.6	0.4
1990	Off	623.5	444.5	508.1	219.8	1,673.5	0.7
1991	Off	1,065.8	323.8	1,116.7	280.6	1,710.9	0.3
1992	On	1,446.0	984.1	1,093.0	410.7	3,367.1	0.7
1993	On	377.3	100.0	421.4	221.4	663.9	0.3
1994	On	817.7	474.3	566.8	141.9	1,981.4	0.6
1995	On	348.1	192.5	421.2	140.7	910.1	0.6
1996	On	1,010.8	401.3	987.9	533.6	1,947.2	0.4
1997	On	753.2	315.9	567.7	424.2	1,335.6	0.4
1998	On	503.2	111.1	481.7	338.3	1,043.6	0.2
1999	On	616.3	113.7	571.4	419.3	850.0	0.2
2000	On	675.8	266.0	567.3	337.1	1,472.0	0.4
2001	On	923.8	67.9	979.6	838.5	1,003.5	0.1
2002	On	473.7	170.9	432.2	143.5	1,137.8	0.4
2003	On	485.0	110.8	425.4	419.2	835.7	0.2
2004	On	731.4	236.1	569.3	545.7	1,192.5	0.3
2005	On	809.1	227.8	849.5	548.8	1,107.7	0.3
2006	On	574.7	137.4	567.4	421.1	1,011.1	0.2
2007	On	992.7	292.8	1,067.6	426.1	1,841.4	0.3
2008	On	772.3	109.8	791.7	483.3	1,102.7	0.1
2009	On	561.0	99.9	511.3	508.3	1,113.9	0.2
2010	On	570.6	112.3	567.3	422.4	720.7	0.2
2011	On	878.2	209.4	1,008.2	507.9	1,702.6	0.2
2012	On	811.0	244.8	711.3	649.3	1,603.3	0.3
2013	On	751.5	151.0	685.3	676.5	1,787.3	0.2
2014	On	656.2	148.9	623.8	500.6	1,262.6	0.2
2015	On	1,051.3	596.4	866.0	476.1	2,275.6	0.6
2016	On	828.1	530.9	427.1	284.4	1,836.5	0.6
2017	On	980.9	333.6	1,076.9	479.1	1,526.9	0.3
2018	On	848.3	314.4	903.1	424.4	1,328.9	0.4
2019	Off	592.0	263.9	427.7	294.0	1,118.9	0.4
2020	Off	822.9	297.4	779.6	427.3	2,009.5	0.4
2021	On	723.8	253.7	649.2	430.9	1,590.0	0.4
2022	Off	746.7	298.8	680.3	284.9	1,764.7	0.4
2023	Off	641.7	266.7	499.7	283.1	1,308.4	0.4

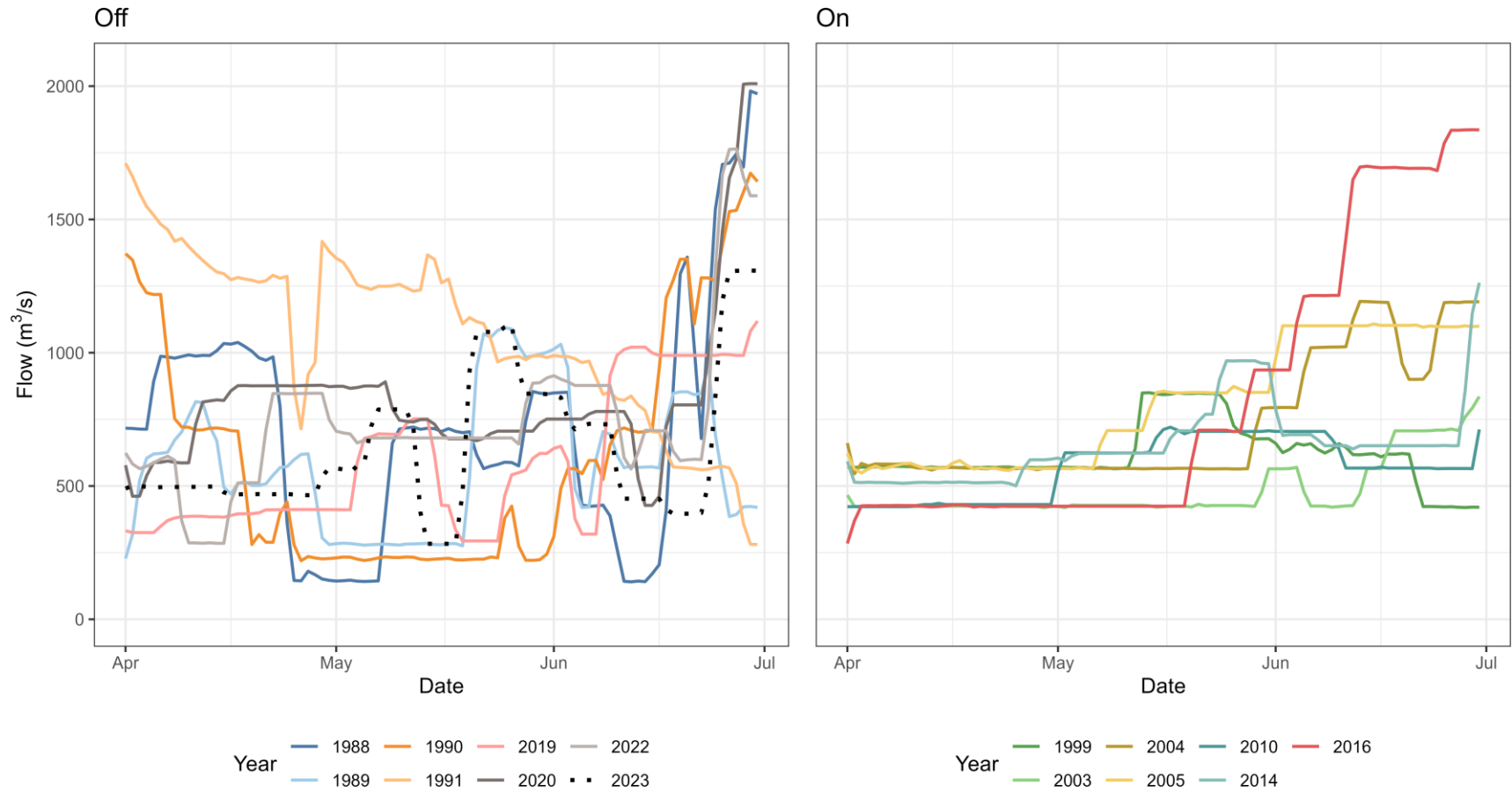


Figure 4-4: Mean daily discharge from HLK_ALH for OFF and ON RBTSPF years used in the productivity models. Experimental year 2023 (black dotted line) was managed as an OFF year.

4.1.2 Mean Daily Water Levels

Level loggers were reinstalled in 2022 at the same locations as previous years, and the last data download occurred in early October of 2023. The level logger at WQIS1 malfunctioned soon after deployment and recorded erratic data (Figure 4-5). In addition, the data from the Kootenay River site (WQC2) could not be safely downloaded in October 2022 due to low water levels that prevented access to the logger by boat or land. The other level loggers at WQIS2-5 appeared to collect accurate elevation and temperature data throughout the deployment period between March and October. Mean daily water levels during the RBTSPF periods were lower in 2022 and 2023 than previous years in the LCR and Kootenay rivers; likely driven by the cool spring with minimal rainfall.

4.1.3 Mean Daily Water Temperature

Temperature data at WQIS1 was more consistent than the elevation data. Mean daily water temperatures varied seasonally ranging from approximately 4 to 17.5 °C at LCR sites between April and October across years. In 2022 mean daily temperatures were generally lower than previous years in the LCR and Kootenay rivers, however, in 2023, recorded temperatures were higher than average (Figure 4-6). Temperatures during the RBTSPF period in 2022 and 2023, steadily increased from approximately 5 to 15 °C in both, the LCR and Kootenay River stations.

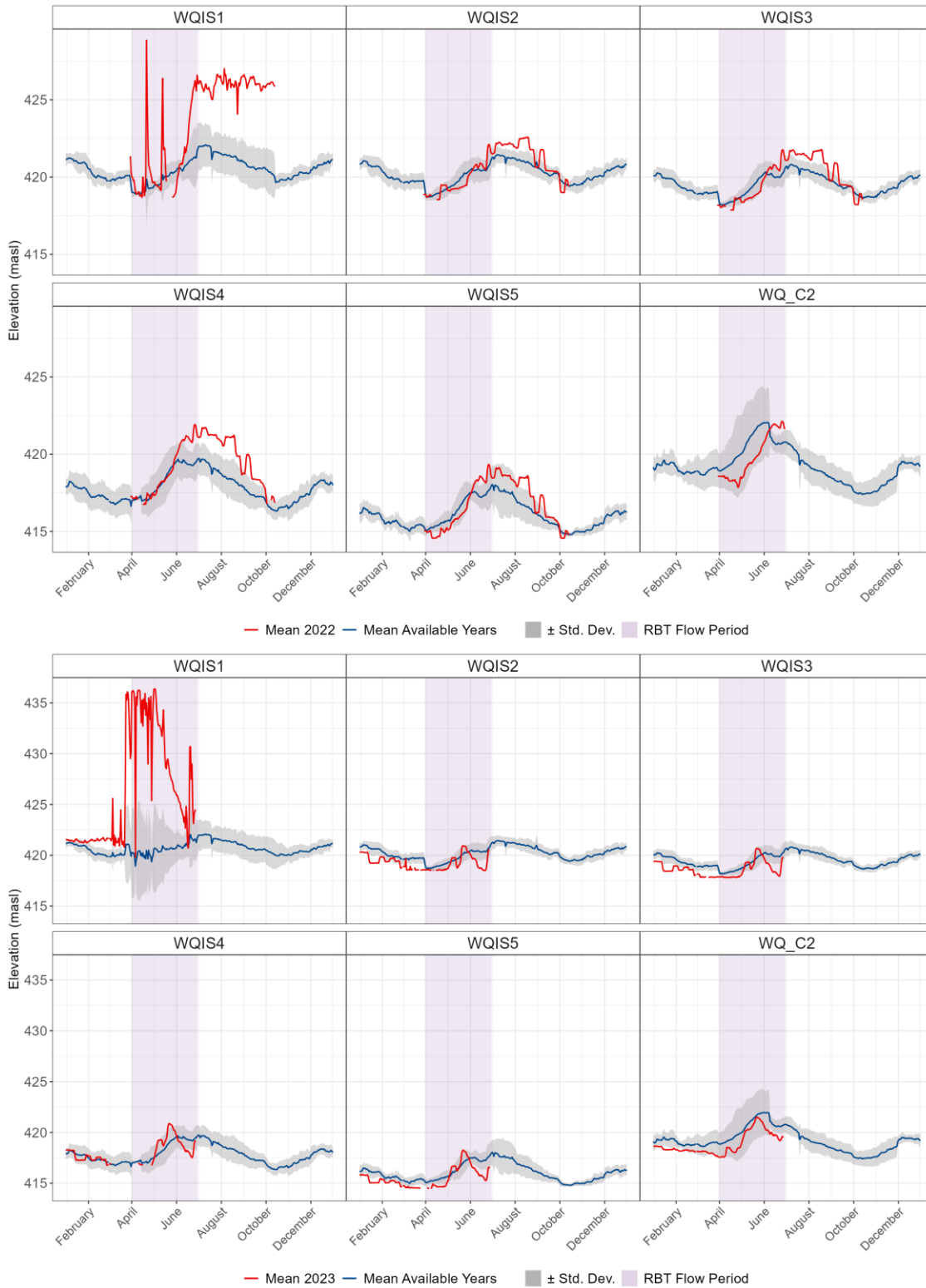


Figure 4-5: Mean daily water levels recorded at WQIS1 – 5 on LCR and at WQ C2 on Kootenay River (Top: 2002, Bottom 2023). WQIS1 malfunctioned during RBTSPF, and data is not used. The blue line is the mean daily water level for 2008 – 2020 and 2022-2023 for LCR sites, and for 2011-2020, and 2022-2023 at the Kootenay River site. Standard deviation is shown in gray.

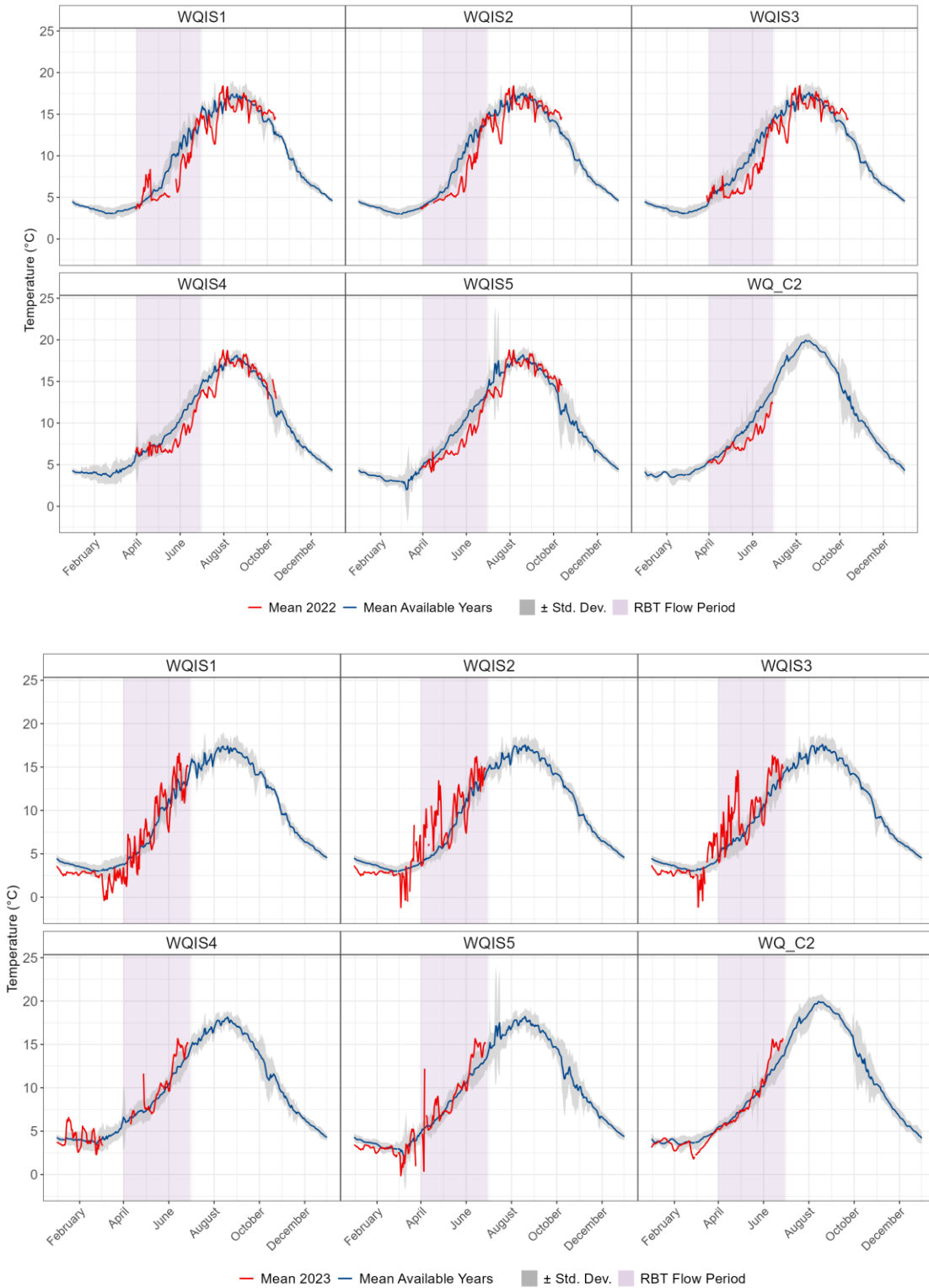


Figure 4-6: Mean daily water temperatures recorded at WQIS1 – 5 on the LCR and at WQ C2 on the Kootenay River. The red line depicts the mean (Top: 2022, Bottom: 2023). The blue line is the mean daily throughout the duration of the study and the gray area shows the standard deviation.

4.1.4 Light Attenuation

The modelled euphotic zone depth was 12.1 m - 95% CI 10.6 - 14.2 m (Figure 4-7). The addition of 2022 PAR and turbidity data confirmed previous analyses and reduced model uncertainty. The Norns Creek fan and the control site R2-S1 were always within the photic zone and receive adequate light to support primary production, in accordance with benthic productivity modelling assumptions. The maximum depth during the RBT flow period of the previously modelled years was 5.5 m on June 29, 1990 (Plewes et al. 2020).

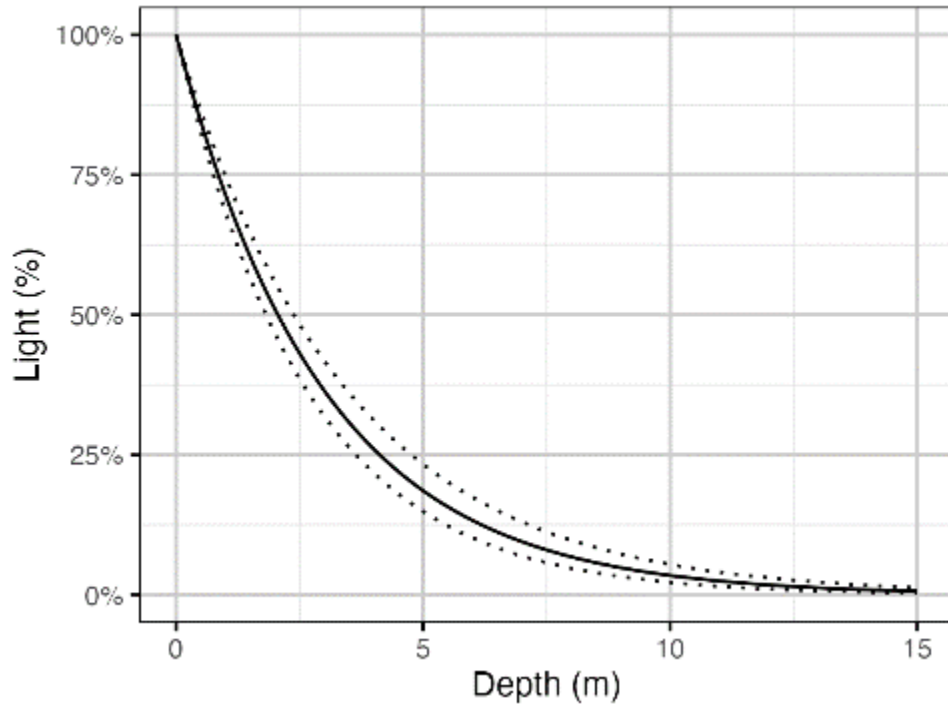


Figure 4-7: Modelled light attenuation by depth based on PAR profiles from the LCR. Dotted lines represent 95 % confidence intervals.

4.2 Benthic Productivity Modelling

4.2.1 Chlorophyll-a

Estimated chl-a accrual during RBTSPF period for both OFF years, 2022 and 2023, was 22 and 10.8 kg for NC fan, respectively. The mean estimated chl-a accrual in NC fan during RBTSPF OFF years was 15.8 ± 3.91 kg, and 18.5 ± 3.5 kg for ON years (Figure 4-8). The mean chl-a accruals for ON and OFF years were not significantly different (90% C.I. for the true chl-a means, 2-sample t-test, $t = 1.4$, p -value = 0.09). The power of the t-test went from 0.3 to 0.52 after including 2023 simulation results, meaning that uncertainty was reduced and the chance of detecting a true effect increased.

In general, years with more stable flows maintained stable wetted areas, reduced periphyton mortality, and thus had higher estimated chl-a productivity. However, flows and resulting accrual varied by year, even among years of the same flow management type. Years 2020 and 2022 (both OFF years) had flow variabilities similar to many ON years and thus had chl-a biomass accruals as most ON years ($CV = 0.4$). Flows were more stable in 2020 than in most OFF years because it was originally managed as an ON year before changing to OFF flow management. The least productive OFF years were 1989 and 2023 with 10.7 kg because of low daily and minimum flows as well as sudden reductions in discharge that increased dewatering and thus zoobenthos and periphyton mortality.

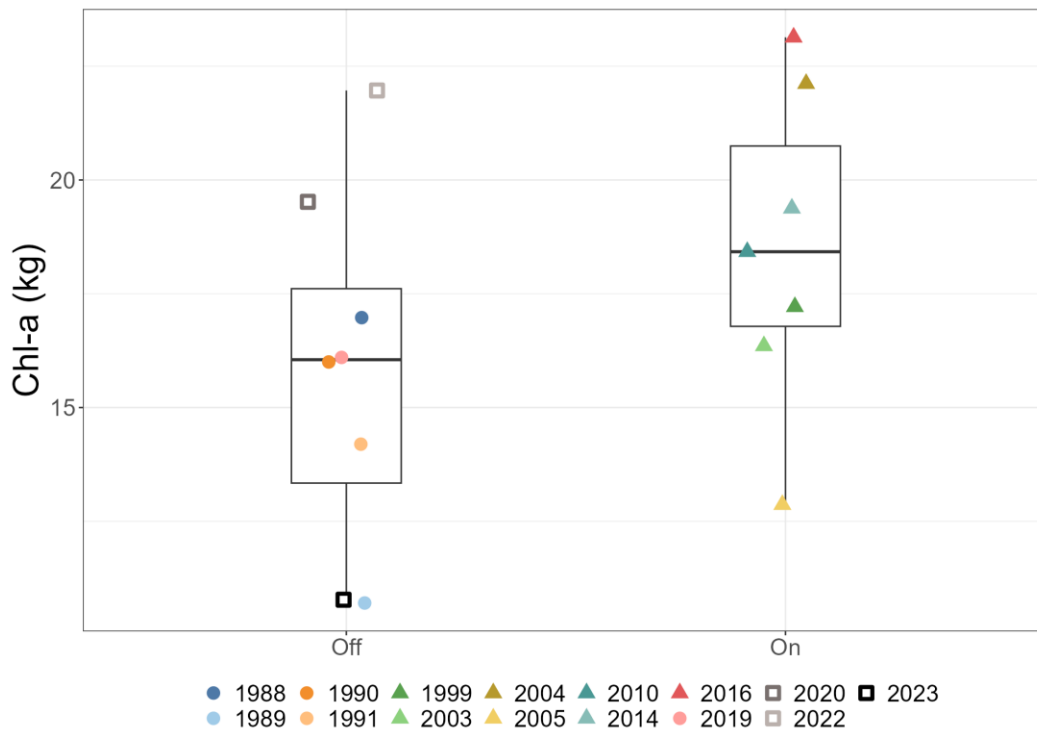


Figure 4-8: Chl-a productivity of Norms Creek fan on June 30th for all simulated years when RBT flows were either ON or OFF. Experimental OFF years 2020, 2022, and 2023 are displayed as open squares.

Estimated chl-a accrual for ON years was highest during 2016 and 2004 with 23.13 and 22.1 kg, respectively. These two years also had relatively high mean flows, as well as the highest maximum

flows of all ON years (Figure 4-4). The OFF years with the highest chl-a biomass accrual were 2022 and 2020 with 22 and 19.5 kg, respectively. These years had relatively high maximum flows, with a maximum of 1,764.7 m³/s in 2022 and 2,009.5 m³/s in 2020.

4.2.2 Invertebrate Biomass

Estimated benthic invertebrate biomass accrual was 279.3 and 156 kg for 2022 and 2023 respectively for the Norns Creek fan during RBTSPF periods. The mean estimated invertebrate biomass accrual for OFF years was 201 ± 47.7 kg and 254 ± 48.5 kg for ON years (Figure 4-9). Benthic invertebrate biomass accrual was significantly higher in ON years than in OFF years (90% C.I. for the true means, 2-sample t-test, t = 2.12, p-value = 0.0269). The power of the t-test increased from 0.6 to 0.7 when adding 2023 results, meaning that, for modelled results, the test has a 70% chance of detecting a true effect.

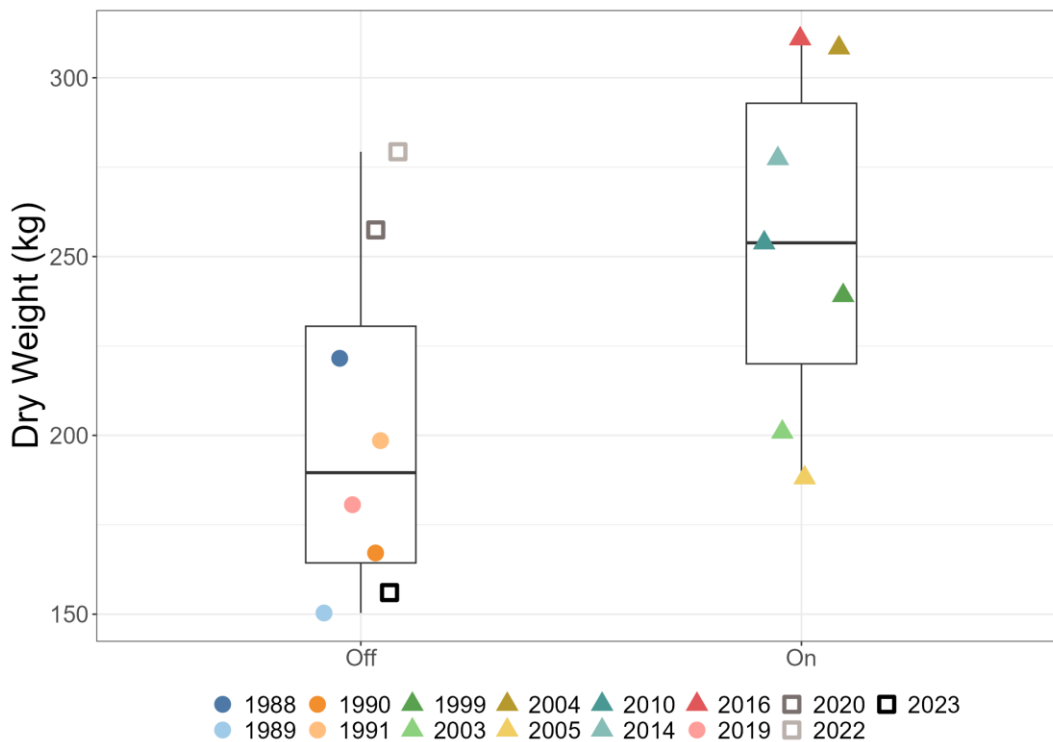


Figure 4-9: Invertebrate biomass (kg) accrual between April 1st and June 30th in all years as predicted by the productivity model. Experimental years (2020, 2022, and 2023) are depicted as open squares.

Like chl-a production, the most productive years for benthic invertebrates were years with the most stable flows. Stable flows increased the available area for invertebrate colonization and reduced mortality due to sudden dewatering, and thus increased invertebrate biomass. The invertebrate biomass accrual for ON years ranged from 188 kg in 2005 to 311 kg. During OFF years, the estimated invertebrate biomass ranged from 150 kg to 279 kg (Figure 4-9). The most productive OFF years were 2020 and 2022, which had relatively high maximum flows of 2,010 m³/s and 1,765 m³/s, respectively. The most productive ON years were 2016 and 2004, which had maximum flows of 1,836.5 m³/s (cv = 0.6) and 1,192.5 m³/s (cv = 0.32), respectively. Years with the lowest estimated biomass also had the lowest flows.

4.3 Measured Benthic Productivity

4.3.1 Periphyton

Periphyton sampler's retrieval occurred in spring 2022 and 2023, while flows were increasing but prior to peak freshet flows. Study results were compared to the larger LCR periphyton dataset. Norns Creek fan samples were dominated by diatoms with smaller contributions by filamentous green, cyanobacteria and flagellated taxa. Productivity metrics from both years indicated that the Norns Creek fan is a productive site in the LCR (Figure 4-10). As it is often the case in the LCR and in large rivers, the most productive periphyton habitat occurred where ideal flow-driven velocities, light, and minimal dewatering intersected.

Norns Creek fan periphyton productivity in 2022 ranged from half to 16 times the historic (years 2008-2014, 2016, 2018, 2022, 2023) chl-a of the control site R2-S1, while 2023 chl-a ranged from 0.2 to 7.9 times the R2-S1 productivity. The 2022 abundance and biovolume ranged from 1.3×10^6 to 9.8×10^6 cells/cm² and 17.2 to 240.4 cm³/m², respectively, which far exceeded the LCR range, while most 2023 Norns Creek fan abundance and biovolume results were similar to the R2-S1 control site range (Figure 4-11).

The most important factor influencing the Norns Creek fan periphyton in spring 2022 was a periphyton bloom of the diatom *Didymosphenia geminata* (Didymo) plus the filamentous green alga *Ulothrix* (Figure 4-12, top panel). All Norns Creek fan productivity metrics far exceeded previously observed LCR values in 2022 due in large part to this bloom and despite it being an OFF year. The Didymo/Ulothrix bloom inflated spring 2022 productivity results because these two taxa have very large cells and because the filament masses created structure for small algae attachment. Both Didymo and Ulothrix are high profile taxa that would be sheared off by high flows, indicating that flows sufficient to shear these filaments had not yet occurred when these 2022 samplers were retrieved. The critical habitat features for these two taxa include full sunlight, stable substrates, and most importantly, stable flows. Didymo mats affect food web structure and ecological functioning (Cullis et al. 2013).

Spring 2023 samples in Norns Creek fan were impacted by an erosion event along the northwest bank of Norns Creek that deposited fines into the fan shortly after the samplers were placed. The fines scoured the periphyton sampler plates repeatedly as the variable flows of the river moved the sand downstream throughout the season. Most plates we retrieved in June were nearly scoured clean, with even the plate numbers scoured away, and higher than usual fines were still noticeable in Norns Creek fan during sampler retrieval (Figure 4-12, bottom panel). This event likely contributed to low periphyton productivity metrics in 2023. Didymo was common in 2023 but not at bloom densities.

Compared to other rivers, abundance, biovolume and chl-a values would place the 2022 Norns Creek fan results in the eutrophic or highly productive category. On the other hand, abundance, biovolume and chl-a values would place the 2023 Norns Creek fan productivity results in the mesotrophic category (Table 4-2). Most LCR samples over the 2008 – 2023 study period would place the LCR in the mesotrophic category.

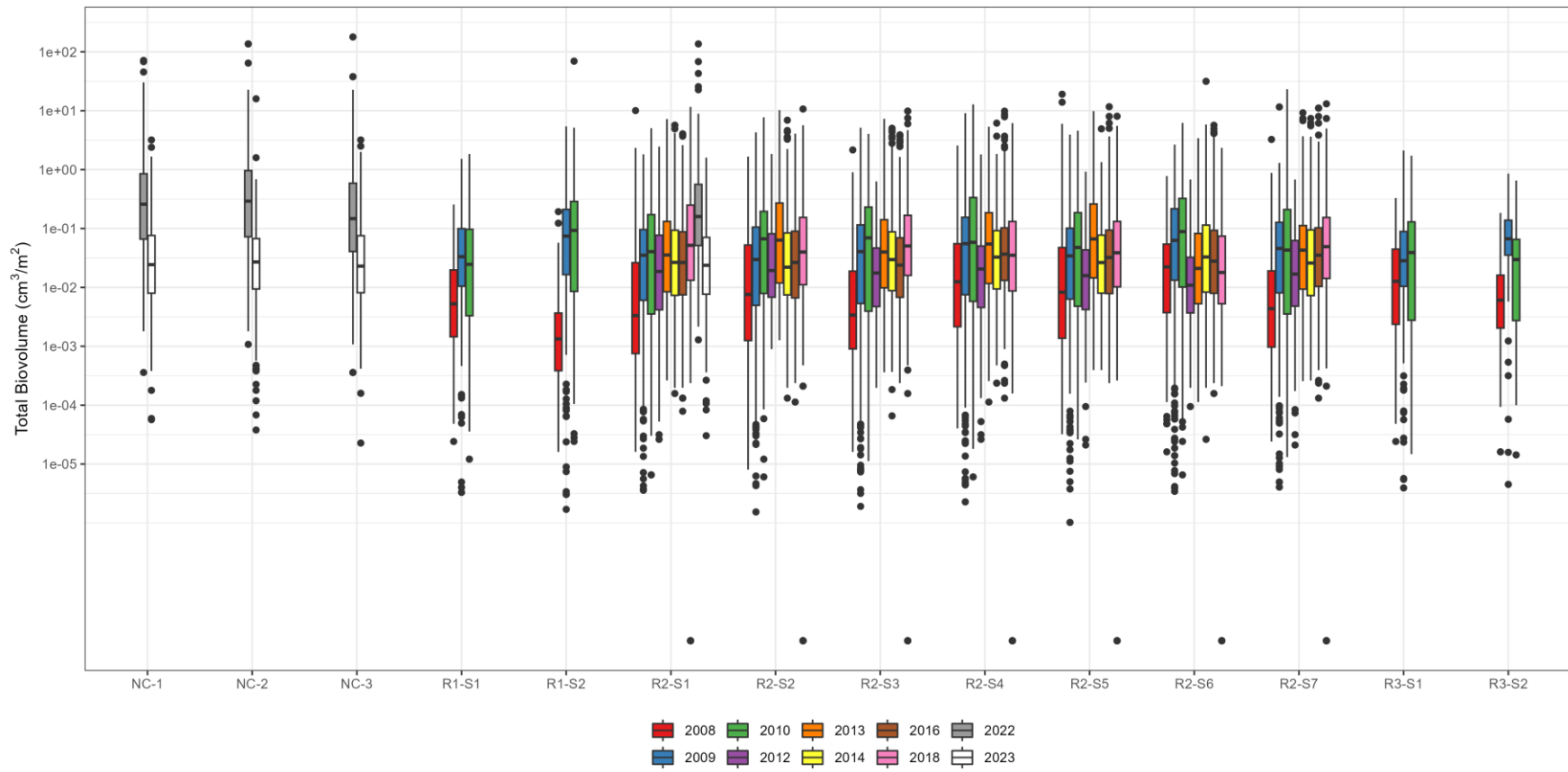


Figure 4-10: Measured artificial samplers periphyton biovolume (log scale) for Norns Creek fan sites (NC 1-3) compared to LCR sites over the full period of LCR studies (2008 – 2023).

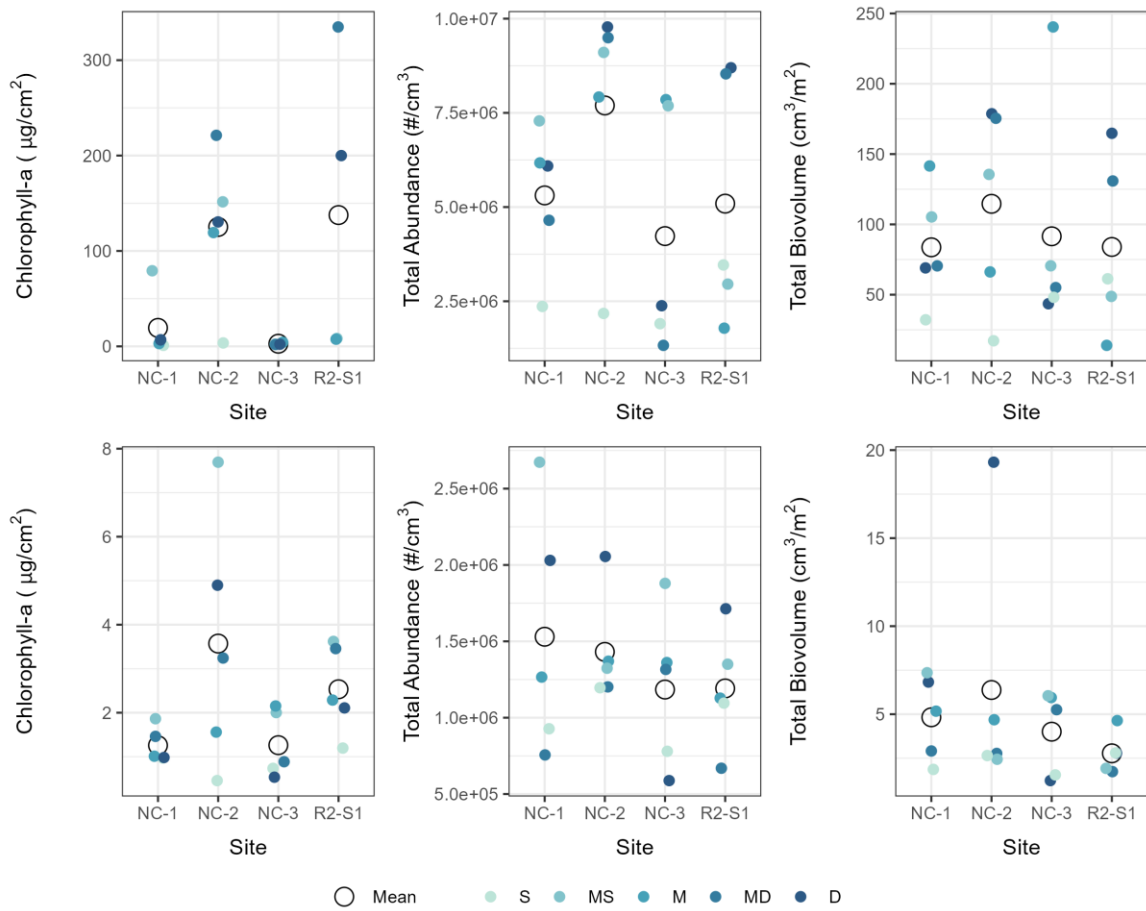


Figure 4-11: Periphyton productivity metrics of chlorophyll-a, total abundance, and total biovolume for Norns Creek fan and R2-S1 (control site) in spring 2022 (Top) and 2023 (Bottom). Site means are indicated with a hollow circle. Notice the different scales for top and bottom panels.

Norns Creek fan periphyton diversity metrics for both 2022 and 2023 (Table 4-3) were in the typical range for large rivers and within the range seen at other LCR sites (Table 4-2). The Norns Creek fan periphyton diversity indices were consistently greater than the adjacent R2-S1 control site data in both years. NC-2 had the highest diversity metrics, including species richness (47), the frequency-based Simpsons Index (0.88), and the true diversity metric, effective number of species (17.6) (Figure 4-13). These results are likely due to a lower rate of exposure of artificial samplers in NC-2, higher average velocities, and substrate.



Figure 4-12: Comparison of periphyton sampler and resultant subsamples retrieved in June 2022 (top panel) versus June 2023 (bottom panel).

Table 4-2: Summary of typical LCR periphyton metrics from 2008 to 2018, and metrics from oligotrophic, typical, and eutrophic systems.

Metric	Oligotrophic or stressed	Mesotrophic	Eutrophic or productive	LCR 2008 – 2018 (median)	Norns Creek Fan	
					Spring 2022 (median)	Spring 2023 (median)
Number of taxa (live & dead)	<20 – 40	25 - 60	variable	8 – 60 (31)	31-51 (42)	29-53 (39)
Chlorophyll-a ($\mu\text{g}/\text{cm}^2$)	<2	2 - 5	>5 – 10 – 30+	0.01 – 55 (3.6)	1.1 – 221 (44)	0.4 – 7.8 (1.5)
Algae density (cells/ cm^2)	<0.2 $\times 10^6$	1 - 4 $\times 10^6$	>10 $\times 10^6$	0.03–3.9 $\times 10^6$ (0.8 $\times 10^6$)	1.3 – 9.8 $\times 10^6$ (6.2 $\times 10^6$)	0.6 – 2.7 $\times 10^6$ (~1.3 $\times 10^6$)
Algae biovolume (cm^3/m^2)	<0.5	0.5 – 5	20 - 80	0.1 – 25 (3)	17 – 240 (70)	~1-19.3 (4.7)

Comparison data obtained from Biggs and Close 1989; Biggs 1996; Durr and Thomason 2009; Flinders and Hart 2009; Freese et al. 2006; Peterson and Porter 2000; Romani 2010.

Diatoms dominated periphyton abundance at all LCR sites in all studied years. Diatoms accounted for more than 90% of Norn’s Creek fan samples and 88% in the control site R2-S1 samples from Spring 2022 and 2023. Of these, the low profile, ubiquitous *Achnantheidium* guild was most abundant in every sample in both years (Appendix C Table A-4). *Achnantheidium* prevalence indicates a variable lotic environment because they are small, rapid re-colonizers. Tiny cyanobacteria accounted for less than 1% of the sample periphyton biovolume in both years (Appendix C Table A-5). Minor amounts of the other algae classes were represented in the Norns Creek fan samples. These algae groups collectively contribute to Norns Creek fan periphyton diversity (Appendix C Figure A-1).

Table 4-3: Diversity metrics in spring 2022 and 2023 for each site.

Year	Site	Total Abundance (#/cm ³)	Taxa Richness (#)	Simpson Index	Shannon Index	Shannon Equitability	Effective Species	Total Biovolume (cm ³ /m ²)
2022	NC-1	5.31e+06	43	0.84	2.62	0.70	14.13	83.69
	NC-2	7.70e+06	47	0.88	2.86	0.74	17.62	114.63
	NC-3	4.23e+06	35	0.78	2.32	0.65	10.18	91.53
	R2-S1	5.09e+06	43	0.86	2.72	0.73	15.35	83.94
2023	NC-1	1.53e+06	38	0.85	2.53	0.70	12.64	4.83
	NC-2	1.43e+06	46	0.86	2.69	0.71	14.88	6.38
	NC-3	1.18e+06	41	0.86	2.59	0.70	13.72	4.01
	R2-S1	1.19e+06	33	0.81	2.37	0.68	11.21	2.78

Didymo was the dominant taxa ranked by biovolume at the Norns Creek fan and R2-S1 in the 2022 bloom, and it was still a dominant in 2023 results, but was less pervasive (Appendix C Tables A-4, A-5). Didymo is endemic in the LCR and is found in most samples at low densities. Over the decade of study in LCR, two winter/spring Didymo blooms were observed, while the other years had much lower Didymo abundance (Appendix C Figure A-2).

The triggers for the 2022 bloom include stable flows with near-bed velocities under 0.2 m/s. Flows that exceed that threshold will shear off filamentous masses (Figure 4-14). Stabilizing flows below HLK may have encouraged the Didymo/Ulothrix bloom observed in Spring 2022 samples. These two algae are high profile taxa that would be sheared off by high flows, indicating that flows sufficient to shear these filaments had not yet occurred when these samplers were retrieved. Regardless of flow management, the intensity of Didymo blooms varied from year to year. Didymo abundance and biovolume did not differ significantly between ON and OFF years in the 2008-2023 data set (Appendix C Figure A-2).

Dominant taxa by biovolume also reflected the importance of large diatoms which are good forage for invertebrates. Large, motile diatoms such as *Cymbella/Encyonema* were prevalent in the sandy substrates of the Norns Creek fan because they can move upward as sand deposits. Epilithic diatoms such as *Staurosira* and *Eunotia* were well represented. Non-motile diatoms from upstream reservoirs were also important in these samples, notably the centric diatoms (*Cyclotella*, *Melosira*), and pennate diatoms (*Ulnaria*, *Tabellaria*). These diatoms are a valuable resource for aquatic consumers.

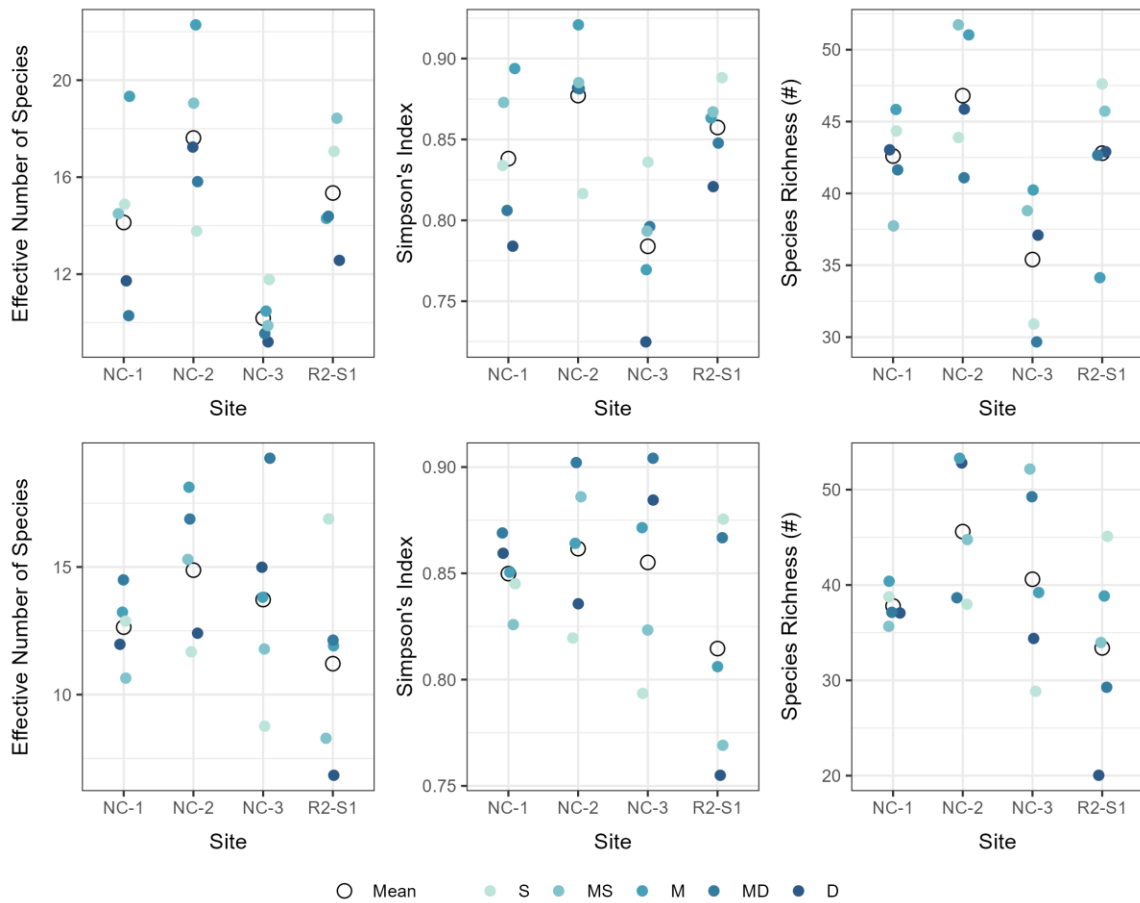


Figure 4-13: Periphyton diversity metrics: Effective number of species, Simpson's index, and Species (taxa) richness, for Norns Creek fan and R2-S1 in spring of 2022 (Top) and 2023 (Bottom). Site means are indicated with a hollow circle.

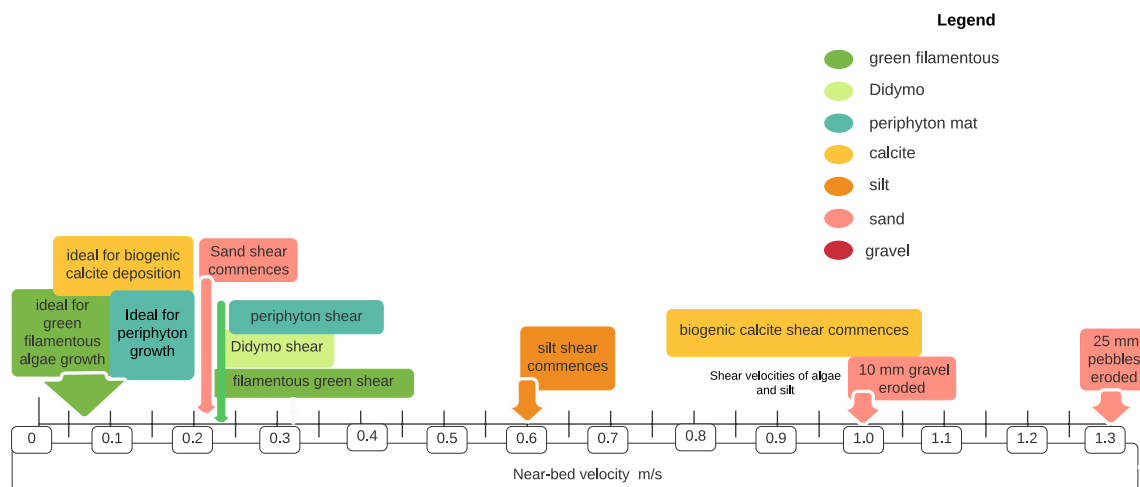


Figure 4-14: Typical shear velocities of periphyton and substrates in streams.

An NMDS analysis conducted on the 2022 data at the family level demonstrated substantial overlap in periphyton community structure between sites ($F = 1.20$, $R^2 = 0.18$, $p = 0.30$) and depths ($F = 1.45$, $R^2 = 0.28$, $p = 0.21$). The 2023 NMDS also demonstrated overlap in periphyton community structure between sites ($F = 0.78$, $R^2 = 0.13$, $p = 0.64$) but sampler depth was distinct ($F = 2.69$, $R^2 = 0.42$, $p=0.008$). As depth increases along each sampler transect, light and dewatering periods decrease. The importance of transect depth may have been obscured by the *Didymo/Ulothrix* bloom in the 2022 data site (Figure 4-15; Appendix C Table A-3). Furthermore, when comparing communities of 2022 and 2023 combined (Figure 4-16), the only factor that is significant is the sampling year ($F = 24.26$, $R^2 = 0.38$, $p=0.001$) when each factor modelled individually with the NMDS vectors as response.

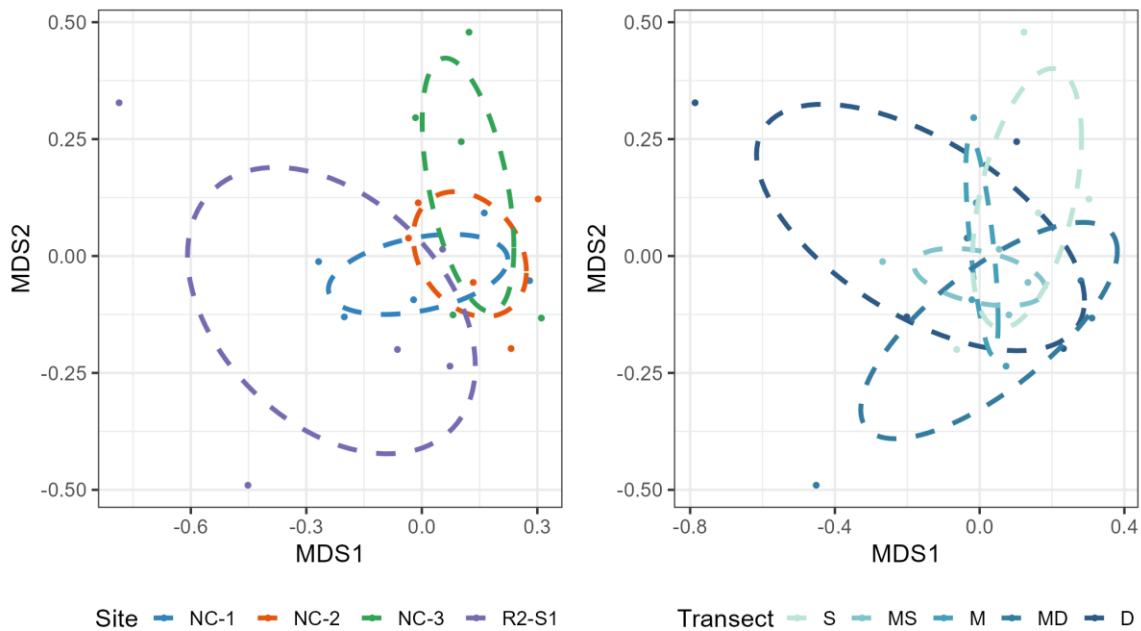


Figure 4-15: NMDS analysis of periphyton community composition by site and transect for spring 2022.

Because each site had a transect of five samplers from shallow to deep, each sample represents a pseudo-replicate. Furthermore, sites in Norns Creek fan were sampled in 2023 for only the second time, so the resulting dataset did not have sufficient statistical power for the previous analysis method of linear mixed effects models.

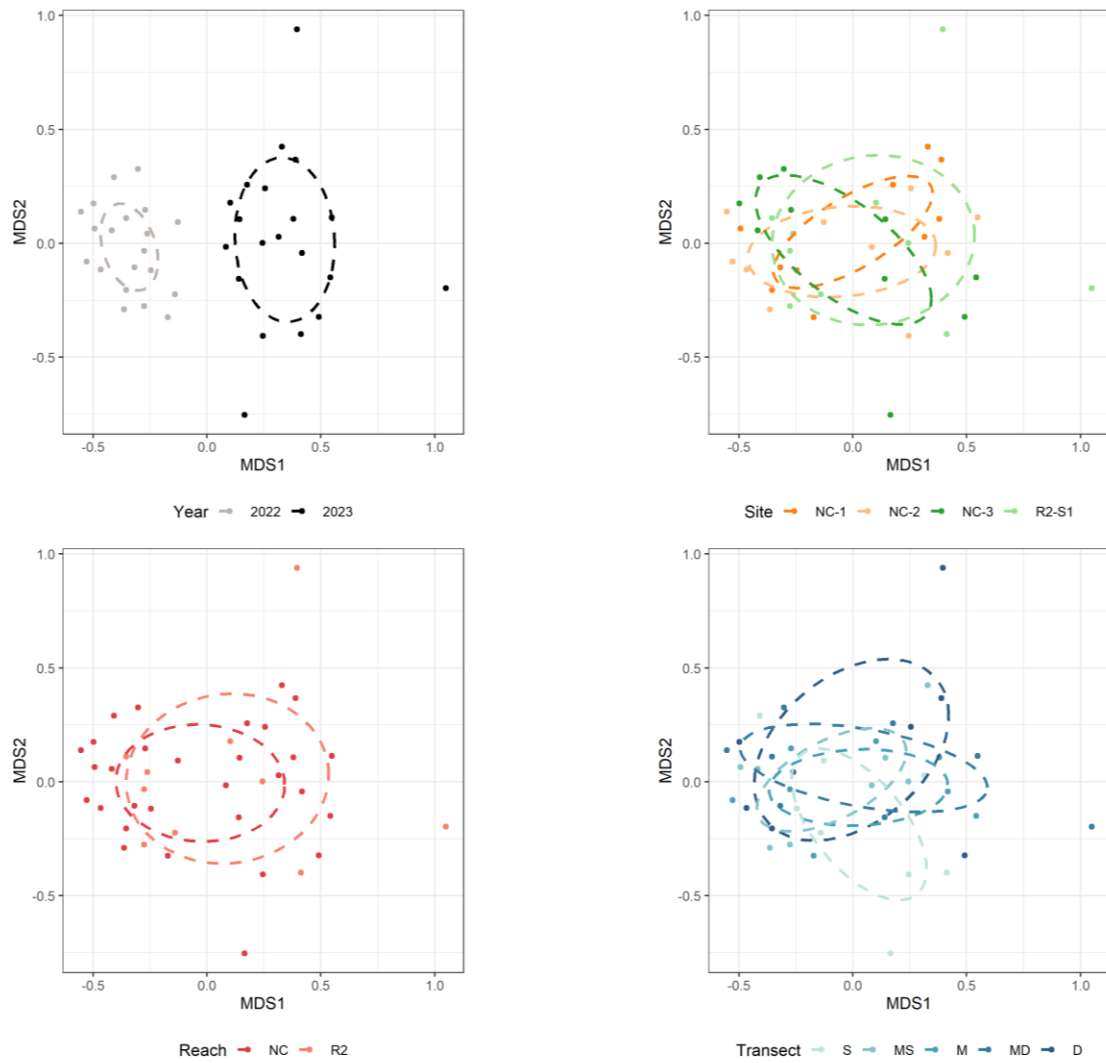


Figure 4-16: NMDS ellipses of periphyton community composition for spring 2022 and 2023 OFF years.

4.3.2 Benthic Invertebrates

The following section presents the spring 2022 and 2023 benthic invertebrate sampling results, community analysis, and comparison with other sites in the LCR that were sampled in previous years and seasons. All previously sampled years were ON years for RBTSPF, while 2022 and 2023 were managed as OFF years.

4.3.2.1 Productivity

Abundance and biomass accrual in NC fan sites dropped in 2023 compared to 2022, contrary to R2-S1 (Figure 4-17) where biomass increased. Mean benthic invertebrate biomass accrual in R2-S1 during the 2022 and 2023 periods was, on average, 7 times higher than the mean invertebrate biomass of the Norns Creek fan sites, 1.4 and .17 g/basket respectively. This difference in invertebrate biomass contrasts with periphyton biomass, where NC fan sites were more productive, driven by periphyton species that do not favor invertebrate biomass. EPT taxa represented, on average, 72 % of the biomass in R2-S1, but only 58 % in NC fan sites. Regardless, spring accruals during the OFF 2022 and 2023 RBTSPF periods (Figure 4-17, left panel) were generally lower than accruals measured in summer and fall, but equivalent to winter accruals for other LCR sites during ON years (Figure 4-18). A similar pattern is observed for abundance numbers for the 2 OFF years, 2022 and 2023, during spring (Figure 4-17, right panel) and the previously surveyed ON years (Figure 4-19).

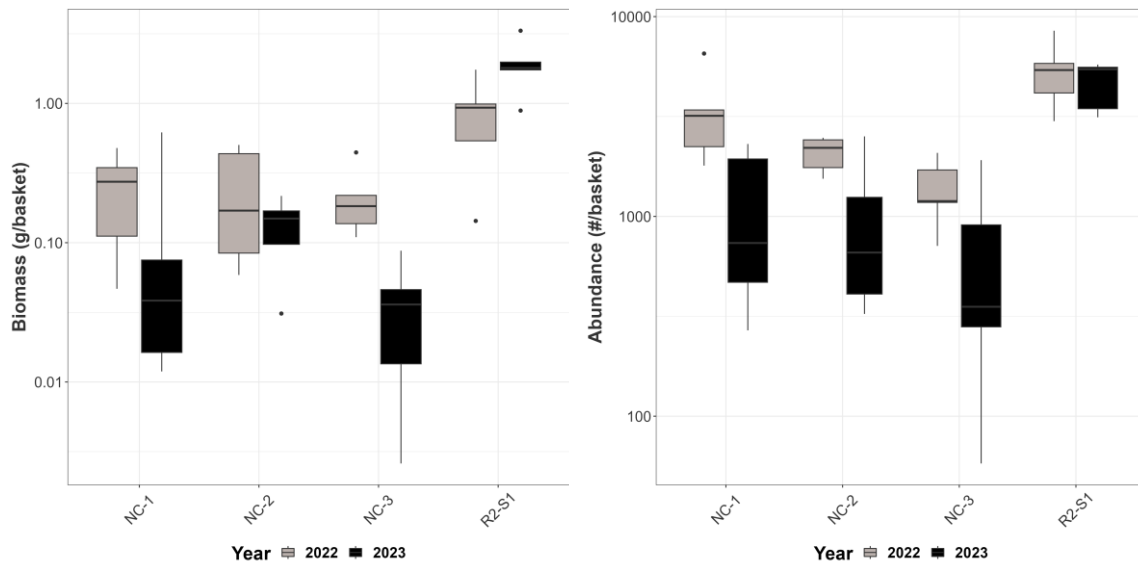


Figure 4-17: Benthic invertebrate dry biomass (left) and abundance (right) (logarithmic scales) measured at Norns Creek fan and R2-S1 in spring of 2022 and 2023 OFF years.

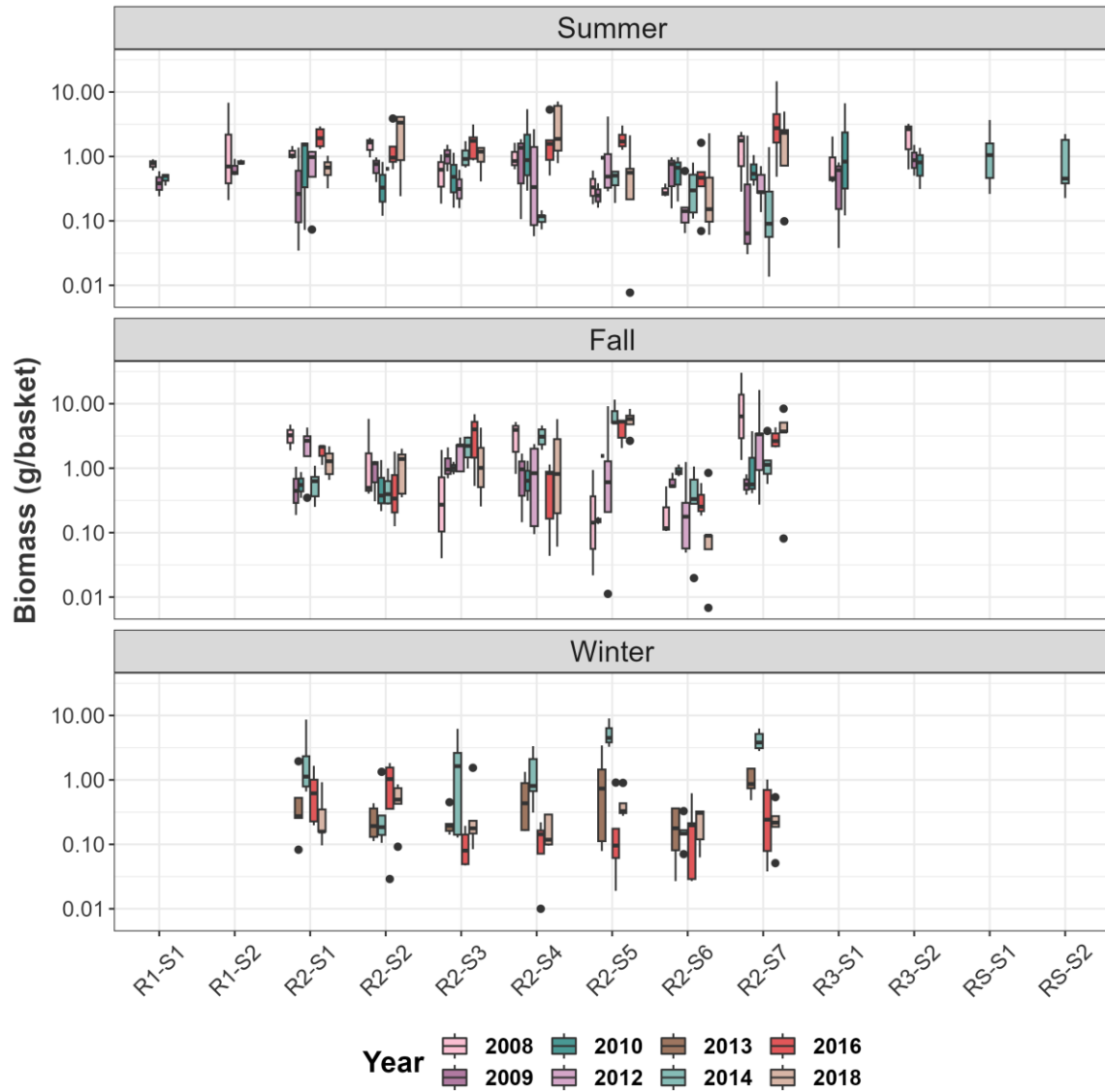


Figure 4-18: Benthic invertebrate dry biomass (log scale) of previously sampled ON years corresponding seasons for each sample site, excluding Norns Creek Fan sites.

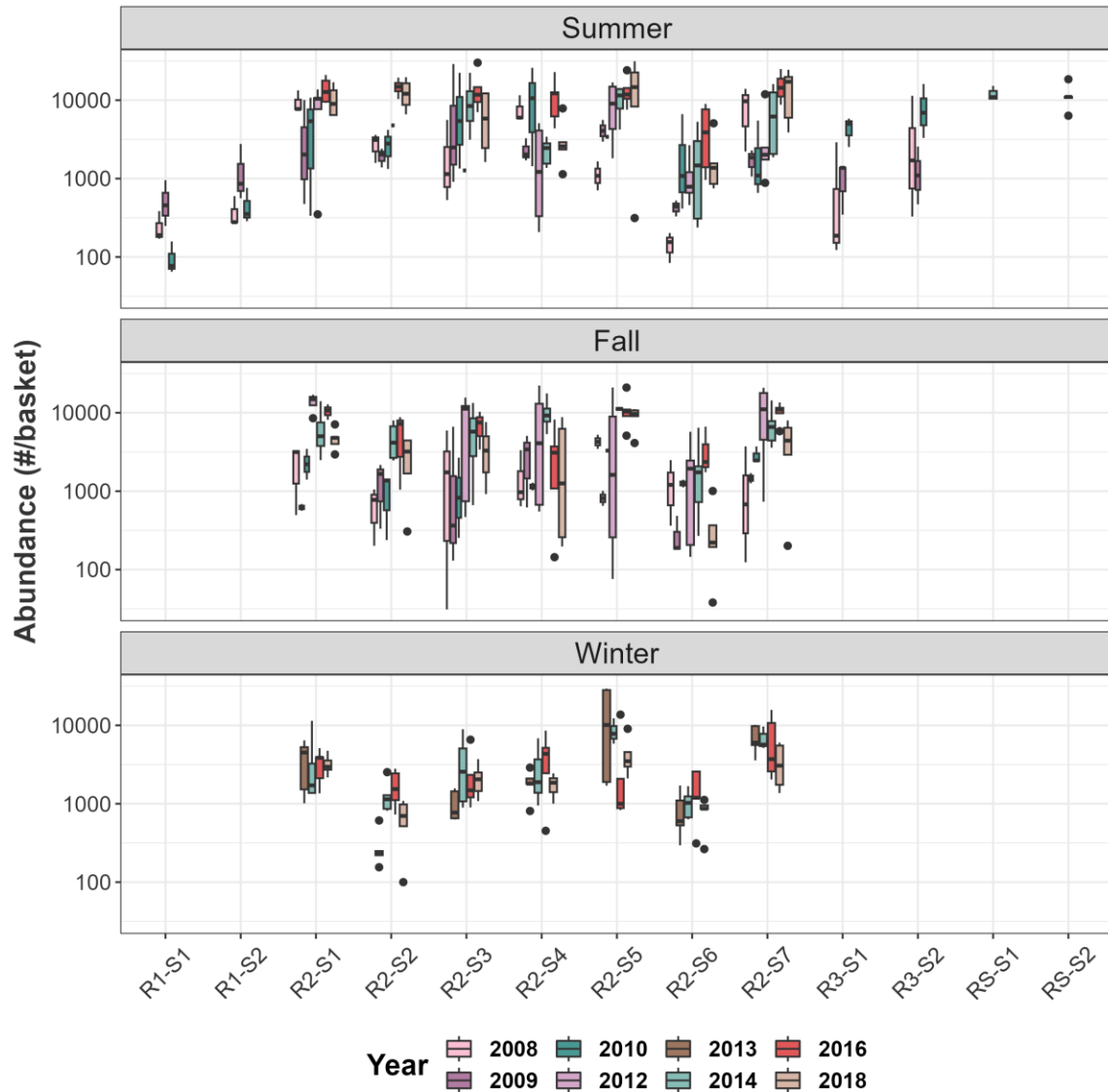


Figure 4-19: Benthic invertebrate abundance (log scale) of previously sampled ON years corresponding seasons for each sample site, excluding Norns Creek Fan sites.

4.3.2.2 Community Composition

While invertebrate productivity dropped from 2022 to 2023, observed diversity increased for all sampled sites. Taxa richness was higher in 2023 (25.7) compared to 2022 (22). NC sites had higher overall taxa richness, but lower EPT richness than the control site (Table 4-4).

Table 4-4: Community diversity metrics by site aggregated for OFF years 2022 and 2023 spring sampling.

Year	Site	Taxa Richness	EPT Richness	Percent EPT Biomass	Effective Species (Diversity)
2022	NC-1	19.0	4.8	73.5	6.2
	NC-2	24.2	5.2	64.5	9.4
	NC-3	23.8	6.6	76.4	9.7
	R2-S1	20.8	8.2	80.1	10.2
2023	NC-1	26.8	6.2	62.5	6.9
	NC-2	27	6	67.8	6.1
	NC-3	25.4	3.4	26.6	6.9
	R2-S1	23.6	9.6	65.1	10.5
Long Term	All	21.7	6.1	47.2	7.7

The dominant taxa (family) at almost all sites in 2022 and 2023 during RBTSPF sampling periods were *Chironomidae* followed by *Naididae* and the mayflies *Ephemerellidae* (Figure 4-20). Despite the community dominance by *Chironomids* and *Naididae* in the Norns Creek fan, biomass contributed by EPT taxa accounted for a mean of 62 % (± 18 %). Regardless of these similarities, community assemblages were different between the Norns Creek fan sites and the R2-S1 control, as well as the other LCR sites (i.e., S2-S7) sampled previously. *Chironomidae* has been the dominant taxa in all sites except S5 and S7 across all years and seasons sampled (Table 4-5). While *Naididae* worms were the second dominant taxa in Norns Creek fan sites, R2-S1 and all other downstream sites in previous studies demonstrated codominance or dominance of *Trichoptera*, notably the net-spinning caddisfly (*Hydropsychidae*). Unlike the Norns sites, *Naididae* accounted for less than 5% relative abundance at the other sites downstream, which had blackflies (*Diptera*, *Simuliidae*) representing the third most dominant taxa. Norns Creek fan and R2-S1 were sampled during spring in 2022 and 2023 OFF years, while previous benthic studies on the LCR were carried out in summer, fall, and/or winter of ON years (2008 – 2018). This seasonal variation in sampling may explain some of the observed differences in benthic invertebrate communities as reflected in the control site (Figure 4-21).

The different distribution of functional feeding groups is tied to the distribution of dominant taxa (Table 4-5). The presence and relative abundance of different functional feeding groups is an indication of the physical habitat attributes that separate each of the sites (e.g., habitat structure heterogeneity, velocity, presence of coarse and fine particulate organic matter, and periphyton growth). Gatherers and collector-gatherers combined to form close to 90% of the community in the NC fan sites, whereas they represented, at the most, 55 % of the community in the rest of the LCR sampled sites (Table 4-6). Invertebrates belonging to these functional feeding groups, *Chironomidae*, mayflies (*Batidae* and *Ephemerella*), *Naididae*, and casemaker caddisflies (*Brachycentridae*), collect fine particulate organic matter from the stream bottom. The control site, R2-S1 had similar composition to NC sites during spring sampling but not in the other seasons. During Summer, Fall, and Winter, R2 sites were dominated or co-dominated by filterers across all sampled years. Filterers collect fine particulate (drift) from the water column. In the LCR, net-spinning caddisflies (*Hydropsychidae*) and blackflies (*Simuliidae*) dominate this group.

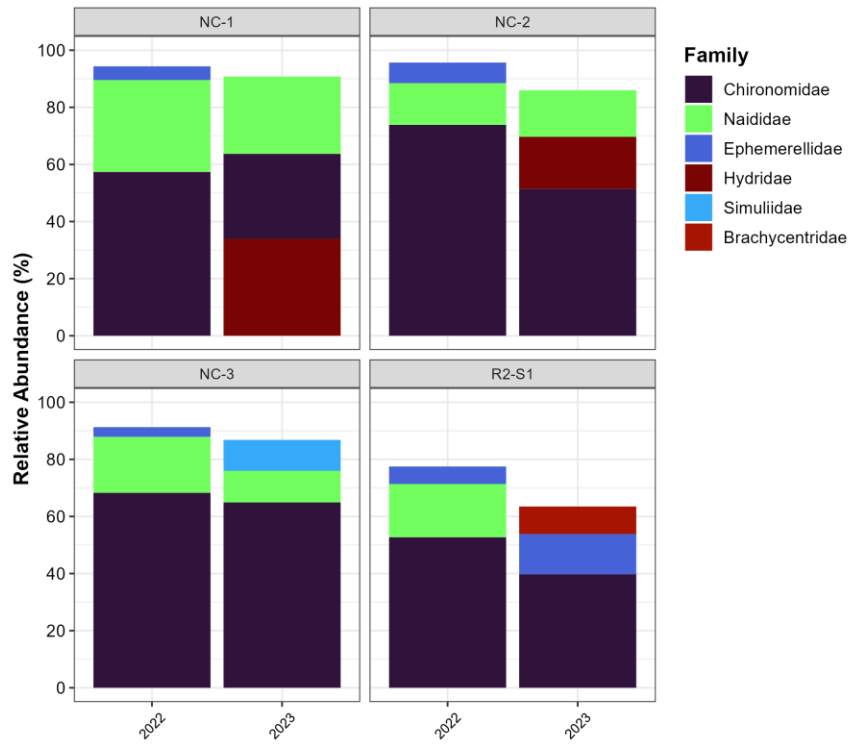


Figure 4-20: Dominant taxa by relative abundance during the RBSPF period in 2022 and 2023 OFF years for Norns Creek fan sites and R2-S1.

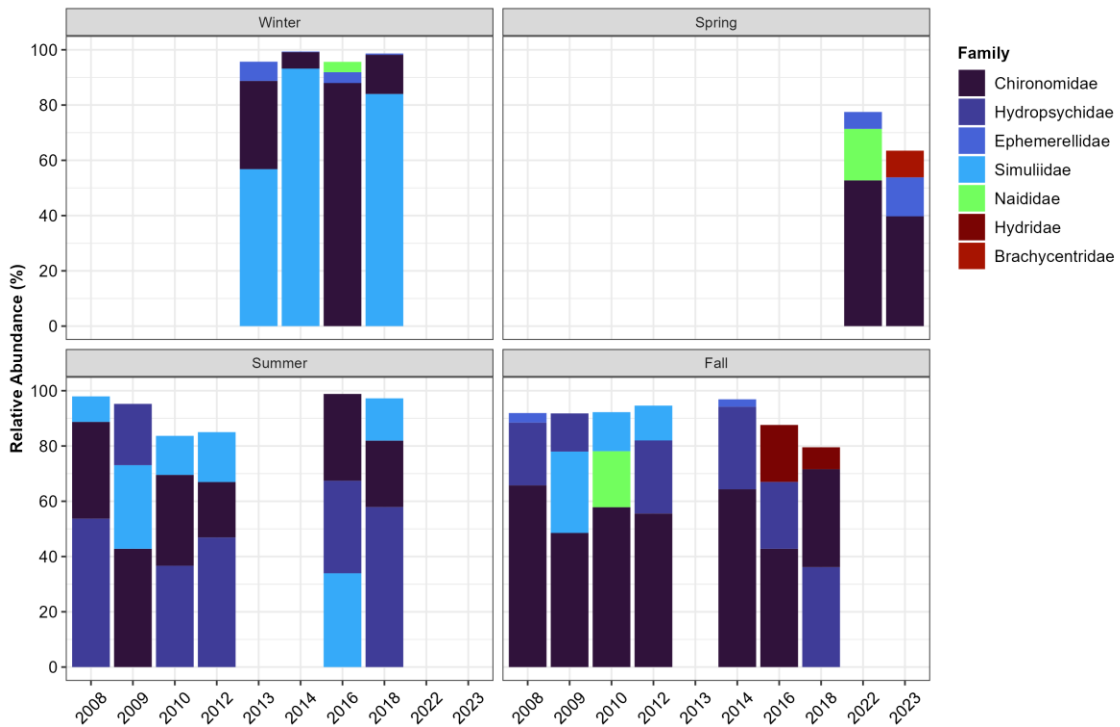


Figure 4-21: Dominant taxa by relative abundance for control site R2-S1 across seasons and years.

Table 4-5: Mean percentage composition of dominant taxa, by abundance, for NC and R2 sites, averaged for all sampled years and seasons. Dominant taxa are shaded.

Taxa Group	Sites									
	NC			R2						
	NC1	NC2	NC3	S1	S2	S3	S4	S5	S6	S7
Percent <i>Chironomidae</i>	47.0	64.2	62.4	42.1	52.1	40.9	38.6	27.5	51.2	30.7
Percent <i>Naididae</i> (<i>oligochaete</i> worm)	31.2	17.4	20.9	3.4	1.3	1.0	3.7	0.6	5.0	2.5
Percent <i>Ephemeroptera</i>	3.5	5.9	2.7	4.1	3.9	5.5	8.0	4.2	4.9	7.0
Percent <i>Trichoptera</i>	2.0	2.1	2.6	24.6	26.1	37.1	29.0	39.8	17.5	35.8
Percent <i>Diptera</i>	2.9	2.3	4.2	23.2	5.4	6.1	11.1	23.5	4.2	17.5
Percent <i>Gastropoda</i>	0.0	0.0	0.1	0.4	4.6	5.8	7.6	3.3	3.3	4.5
Other Taxa	13.3	8	7	2.1	6.6	3.7	2.0	1.0	14.0	1.9

Table 4-6: Functional feeding group mean relative abundance for Norns Creek Fan and R2 sites across all years and seasons.

Functional Feeding Group	Sites									
	NC			R2						
	NC1	NC2	NC3	S1	S2	S3	S4	S5	S6	S7
Gatherer	44.7	40.4	39.6	34.3	48.4	39.0	37.5	23.0	54.9	30.4
Collector-Gatherer	36.7	47.2	45.9	10.3	5.5	5.9	10.8	6.9	6.9	8.6
Filterer	4.2	3.8	6.4	49.4	32.1	43.5	40.1	64.4	18.5	53.3
Predator	13.3	7.7	6.6	2.1	6.7	3.6	2.0	1.1	11.0	1.7
Scraper	0.1	0.0	0.1	0.4	4.8	5.9	7.9	3.4	3.8	4.7
Shredder	0.1	0.1	0.0	2.3	2.1	1.8	1.1	1.0	1.4	0.9

4.3.2.3 Community Structure

The community composition differences previously described for 2022 and 2023 OFF years were also observed when exploring community structure using NMDS. This analysis showed that invertebrate communities, at the family level, differed significantly between Norns Creek fan sites and R2-S1 ($R^2 = 0.2$, $F = 9.74$, $p = 0.001$) (Figure 4-22). Communities within the same site or reach did not differ by transect ($R^2 = 0.05$, $F = 0.47$, $p = 0.964$) for 2022 and 2023.

The structure of the invertebrate communities changed between 2022 and 2023 RBTSPF sampling periods. *Chironomidae*, *Enchytraeidae*, and *Ephemerellidae* contributed the most to the NMDS vectors in 2023; while *Amphipoda*, *Trichoptera*, *Limnephilidae* did it in 2022, mainly because the three dominant families were the same across sites in 2022, *Chironomidae*, *Naididae* and *Ephemerellidae*, but changed across sites in 2023 (Figure 4-20). Community structure similarity reflected the differences between 2022 and 2023. Clusters followed the differences observed between the two years when combining both samples (Figure 4-23 – Bottom panel), while family assemblage differences are more clear in a per-year clustering (Figure 4-23 – Top panels). Most influential species between groups in 2022 were similar between clusters: *Chironomidae* and

Naididae. On the other hand, the influence of these 2 families was lesser in 2023, with *Hydropsychidae* and *Ephemerellidae* being also influential.

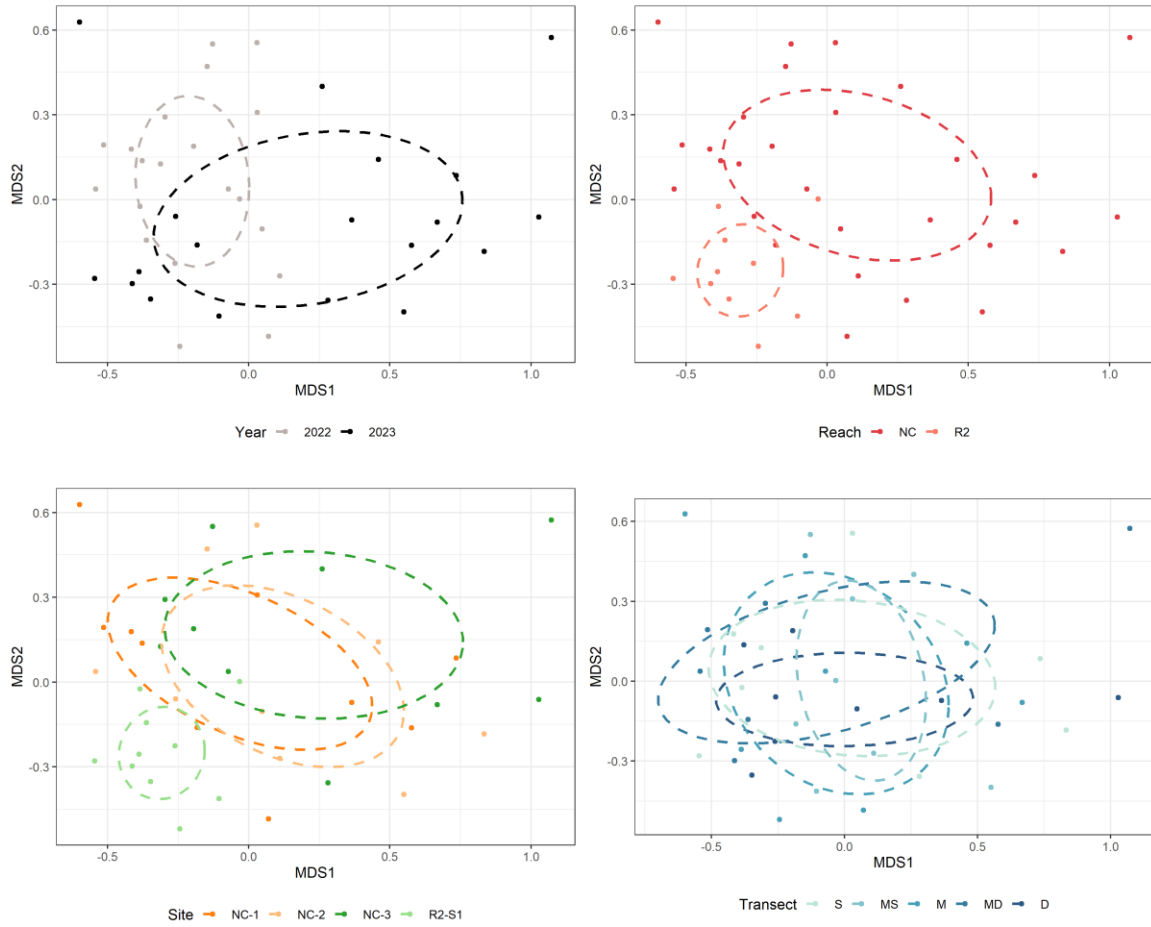


Figure 4-22: NMDS ellipsis plots for benthic invertebrate community composition by year, reach, site, and transect for OFF RBTSPF period samples in 2022 and 2023. Overlapping ellipsis indicate similarity.

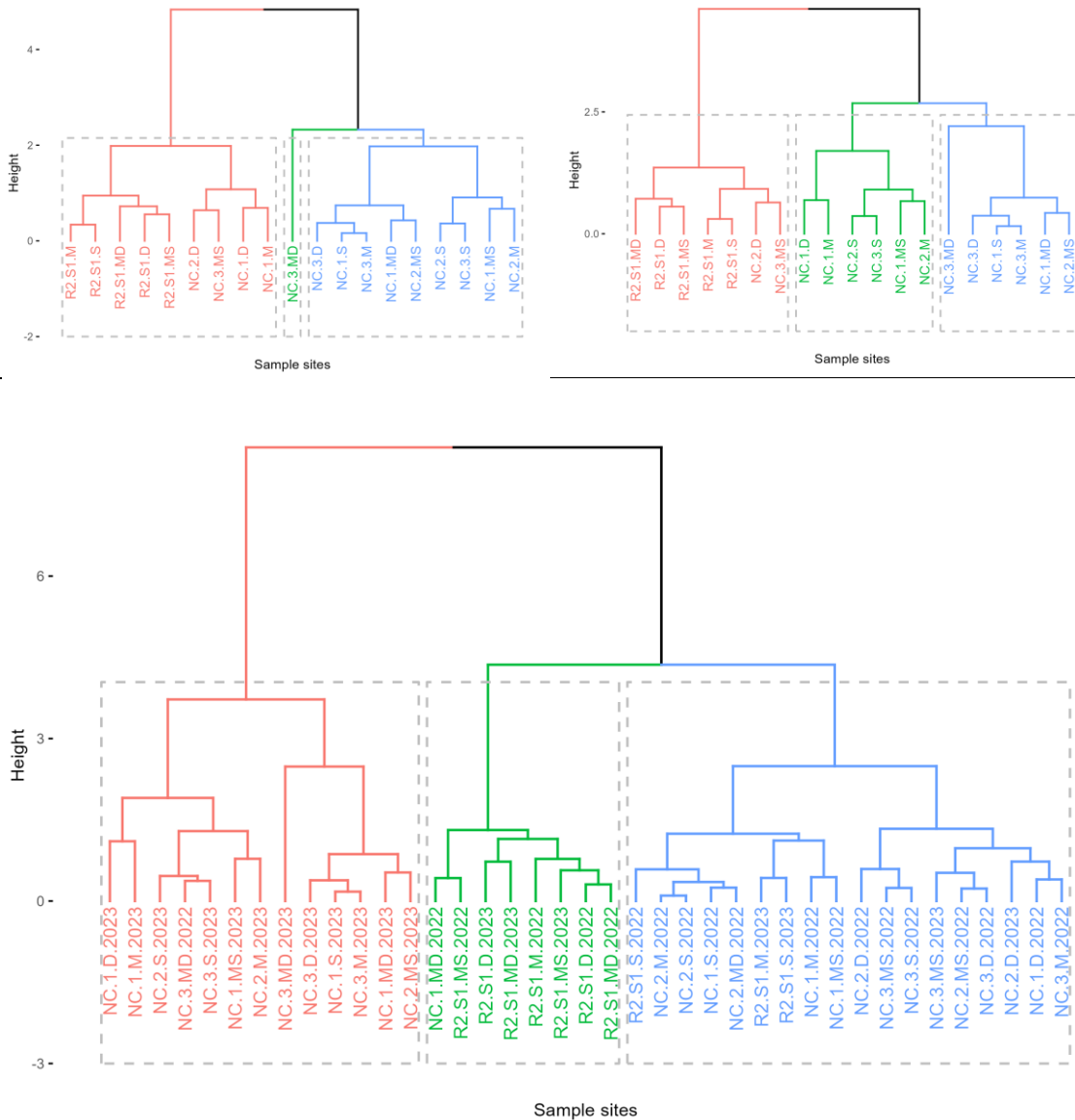


Figure 4-23: Cluster plots for benthic invertebrate community composition for 2022 (Top left), 2023 (Top right) and 2022 - 2023 combined (Bottom) RBTSPF period samples. Branches indicate dissimilarity.

Community structure across years also showed differences when comparing 2022 and 2023 OFF years spring sampling with the communities of previously sampled ON years (2008 – 2010, 2012 – 2014, 2016, and 2018). Keeping in mind that sampling seasons do not match between the two OFF years and the previously sampled years, community structure showed significant differences across all the factors when modelled individually. When community model is grouped by site and the longitudinal aspect of the data is considered, it is the combined effect of year/season that explain most of the observed variance ($R^2 = 0.321$, $F = 2.485$, $p\text{-value} = 0.0014$).

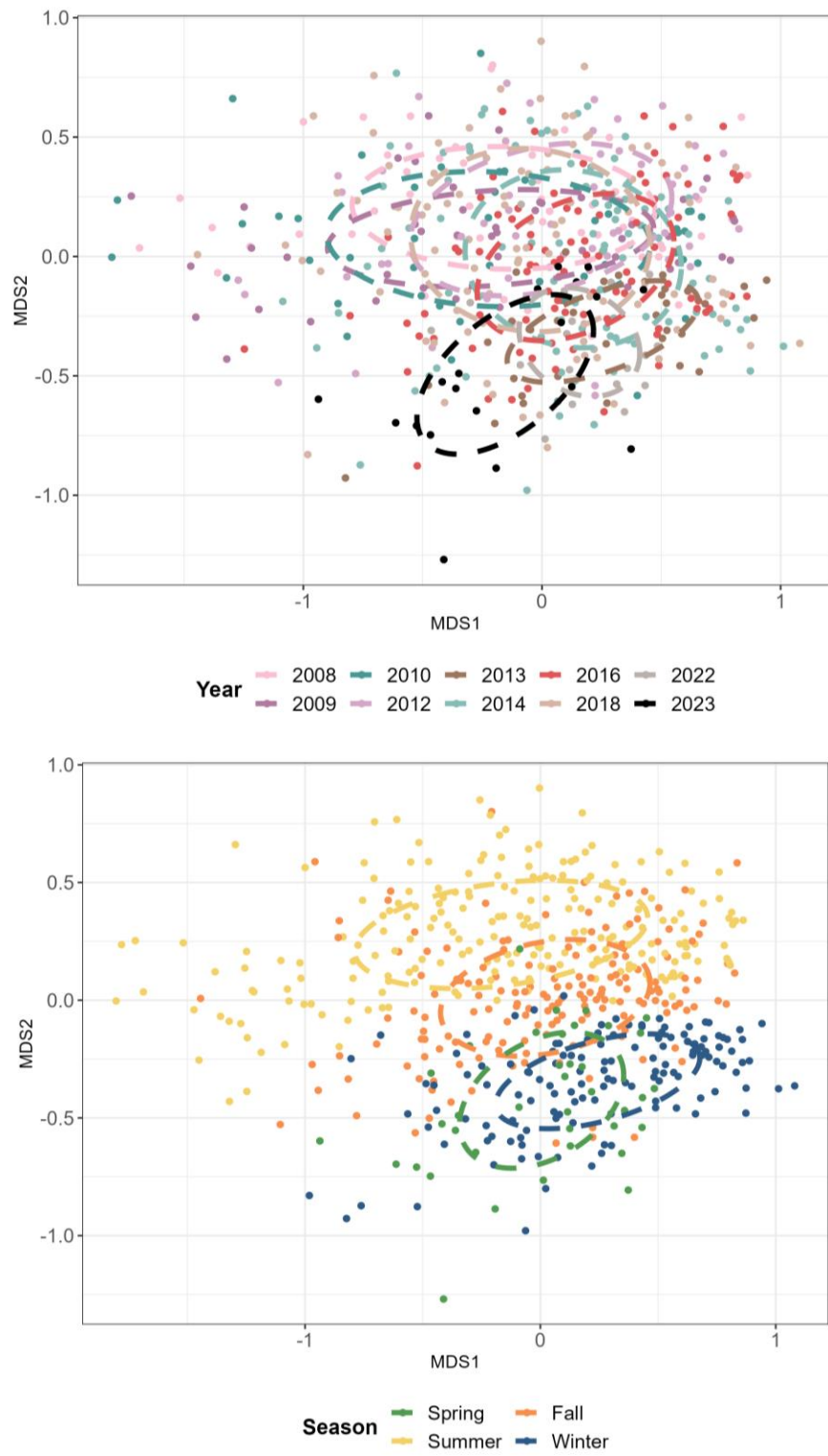


Figure 4-24: NMDS ellipsis plots by year (top) and season (bottom) for benthic invertebrate community composition for all sampled years and seasons data combined.

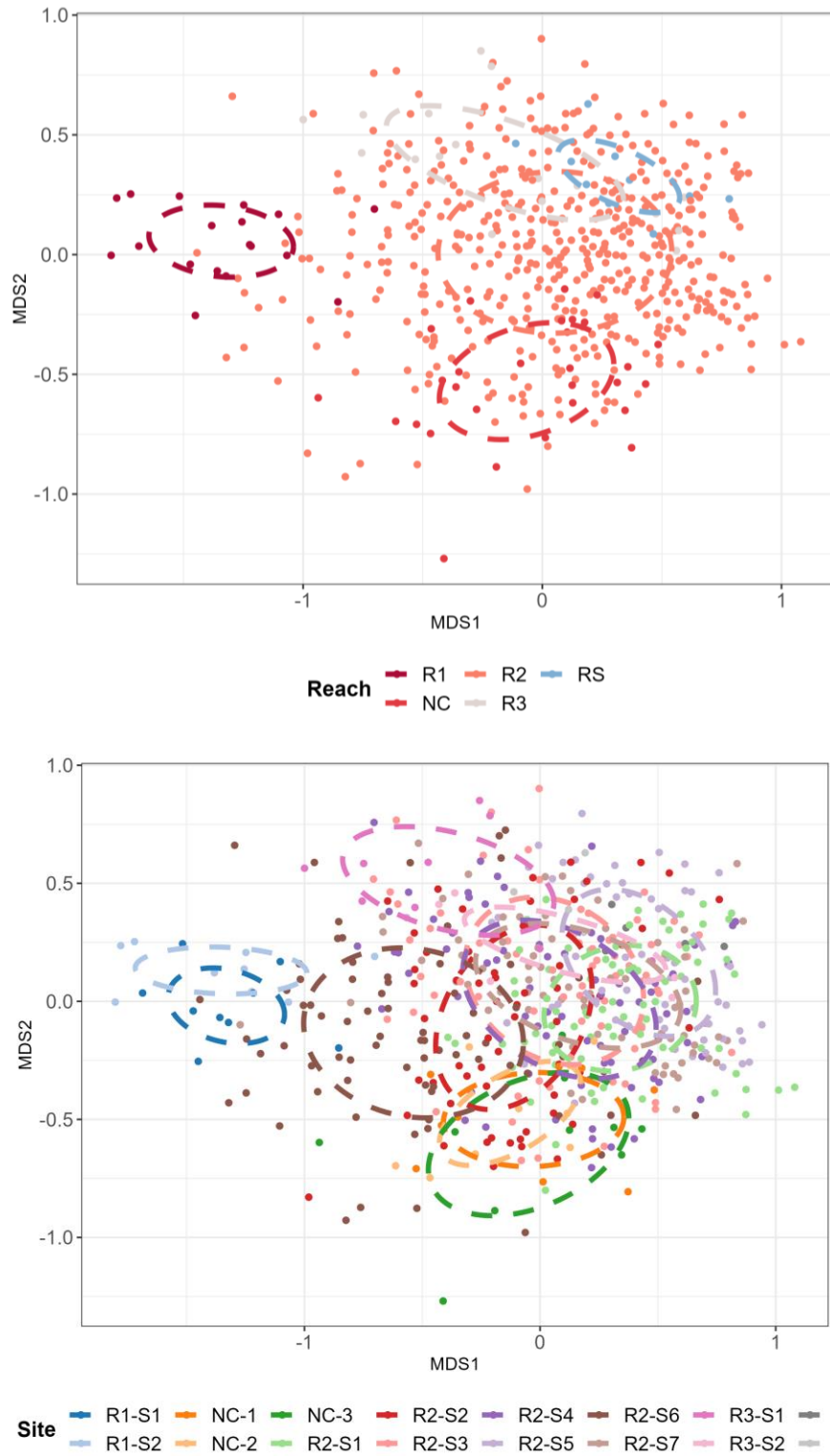


Figure 4-25: NMDS ellipsis plots by reach (top) and site (bottom) for benthic invertebrate community composition for all sampled years and seasons data combined.

5 CONCLUSIONS

OFF and ON years differed in terms of flows coefficients of variation (*cv*), with ON years showing less variability during RBTSPF periods than OFF years. These differences, manifested in terms of water depth, and hence wetted areas, affect productivity downstream of the HLK dam. According to the productivity models, the presence or absence of RBTSPF has more influence on benthic invertebrates than on periphyton. This could be explained because invertebrates are likely more sensitive to substrate dewatering and have slower spring recovery rates than periphyton (Schleppe et al. 2015). The inclusion of 2 additional OFF years in the modelling helped reduce the uncertainty in assessing the effect of RBTSPF on benthic productivity and validate modelling results.

While productivity for invertebrates was higher for ON years, some OFF years, like 2022, with flow regimes similar to ON years had ON-like estimated productivity. The variation in flows inside ON or OFF regimes can be high between years, due to management or environmental conditions. This indicates that other metrics, beside CV and intended RBTSPF management goal could be used to characterize flow patterns. Benthic production reflected these patterns in flow, with relatively stable flows in 2022 that created higher estimates of benthic productivity compared to other OFF years and similar to estimates of ON years. Accrual is solely based in area submergence/exposure response. While substrate submergence is a strong predictor of productivity, increased stability of flows and increased minimum flows also increase productivity and benthic invertebrate diversity (Ellis and Jones 2013, Malmqvist and Englund 1996, Plewes et al. 2020).

The differences in estimated productivity could not be fully validated with empirical data as the number of sampled ON (8) and OFF (2) years did not provide enough consistent replicates due to unbalanced number of years and seasonal sampling. Additionally, exceptional events during sampling years 2022 and 2023 might obscure the effects of RBTSPF. Primary production in 2022 was influenced greatly by a large *Didymo* bloom. Together with green algae, these filamentous taxa inflated the periphyton productivity metrics to levels not previously seen in the LCR.

NC fan sites had, on average, lower benthic invertebrate biomass and EPT richness than other LCR sites. These values indicate that the productivity of NC fan, after emergence, may not be as important to Rainbow Trout (RBT) populations as the production of other habitats in the LCR. The smaller substrate limited interstitial habitat, and low instream cover of Norns Creek fan, are not ideal conditions for RBT rearing. Juvenile RBT likely leave the fan and move to better foraging habitat after emerging from the substrates.

Invertebrate communities tend to be dominated by one or two taxa with a large number of rare species in more unstable aquatic habitats (Death, 1996). The benthic invertebrate community assemblage in the LCR is characteristic of a large, regulated river. Chironomidae and Naididae dominate the Norns Creek fan of the LCR, which experiences drawdown and emersion of over 40% of the fan. These taxa are common and important in these varial zone habitats because they are relatively resistant to desiccation. Chironomidae and some Naididae are capable of surviving emersion of more than three weeks under field conditions (Poznańska et al. 2016). Some studies have shown that *Naididae* dominate emersed sediments with the highest densities immediately following re-wetting (Poznańska et al. 2016). For example, some *Naididae* species demonstrate resistance to drying by forming cysts or through behavioral adaptations of vertical migrations deeper into sediments to moisture habitats (Poznańska et al. 2016).

6 RECOMENDATIONS

If simulated results are to be validated against empirical sampling, additional sampling of OFF RBTSPF years would be required. Moreover, given the observed flow variability, it may be advantageous to explore modeling LCR benthic productivity based on additional flow factors. Recent modeling of the Columbia River's benthic productivity for the Columbia River Treaty used a suite of flow parameters as predictors, rather than water level alone (Akers et al. 2021). These predictors included variability of daily flows, number of high flow events, mean duration of high flow events, fall rate of decreasing flows, daily flow magnitude changes, and maximum annual flow (Akers et al. 2021). A modelling approach with similar flow parameters and the inclusion of flow thresholds, similar to the CRT model, may capture the effects of nuances in flows between and among ON and OFF years. This approach could more accurately predict the impact of flow management with finer detail, rather than just comparing between ON and OFF years exclusively. This detail will help refine the modeling estimates to the effects of changes in specific flow characteristics. For example, it could help determine if overall maximum flow, mean flow, or flow variability have more effect on inundated area, and thus chl-a and invertebrate accrual. It may also offer more insight into the specific effects of flow management decisions, which can help inform future LCR flow management. If interest and capacity to improve the productivity model remains, we suggest exploring a format similar to the CRT model.

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APPENDIX A LITERATURE REVIEW

1.1. Literature Review

Benthic invertebrate community assemblages in regulated rivers are influenced by species' abilities to tolerate rapid changes in water level and temperature. Responses to and survival of dewatering are diverse, and vary by species, life history, behaviour, and morphological traits (Jenkins and Boulton 2007, Robson et al. 2011, Thorat and Nath 2018). For example, some species with desiccation-resistant eggs can hatch and recolonize quickly after drought when their eggs are rewatered, so while mortality following a dewatering event can be high, populations recover relatively quickly (Jenkins and Boulton 2007, Miller et al. 2020, Robson et al. 2011). Behavioural responses can also change mortality rates of invertebrates. Some Chironomus larvae show grouping behaviour of bunching together to reduce water loss by evaporation (Thorat and Nath 2018). This grouping behaviour creates a density-dependent response to dewatering, where survival changes depending on how many individuals are present and grouping. Some more mobile species can also move into residual pools of standing water following dewatering to avoid desiccation (Lancaster and Ledger 2015). Some species of Ephemeroptera and Chironomidae can tolerate daily fluctuating flows and desiccation, and these taxa usually dominate the macroinvertebrate community in regulated rivers (Brittain and Saltveit 1989, Munn and Brusven 1991, Poznańska et al. 2016, Walters and Post 2011). For example, one study showed the population densities of several species of Naididae and four taxa of Chironomidae were not impacted by drying events lasting five to six days (Lancaster and Ledger 2015).

Survival of fluctuating flows and recolonization following low flows vary widely by river system and species, even within the same taxa (Brittain and Saltveit 1989). For example, Suemoto et al. (2004) found differences in desiccation tolerance between larval Chironomid species. Specifically, they found that *C. salinarius* tolerated desiccation better than *C. kiiensis* and *C. yoshimatsui*, yet Kawai et al. (2000) found that these two latter species had higher desiccation tolerances than *C. salinarius* (Kawai et al. 2000, Suemoto et al. 2004). The differences in desiccation tolerance between the two studies may be due to varying salinities and the species' varying abilities to tolerate different salinities (Suemoto et al. 2004). These studies demonstrate how physical and chemical habitat conditions will influence species' responses to dewatering and survival of desiccation.

Flow variability in regulated river systems influences community assemblages of macroinvertebrates by creating a selective pressure for species that can survive desiccation or recolonize quickly. Overall macroinvertebrate abundance can increase or decrease with river impoundment, but regulated rivers have lower macroinvertebrate diversity and taxa richness when compared to unregulated rivers or rivers that have more seasonal flow variability (De Jalon et al. 1994, Ellis and Jones 2013, Gislason 1985, Milner et al. 2019, Munn and Brusven 1991, Steel et al. 2018). For example, total abundance and species richness of Ephemeroptera was lower in regulated rivers when compared to unregulated rivers, but increased constancy in flow increased Ephemeroptera richness (Malmqvist and Englund 1996).

Annual variation in flows in regulated systems can also shift invertebrate community composition and impact different taxa and functional feeder groups in different ways. For example, aquatic taxa were more abundant during high flow years in regulated systems, and low flow years reduced aquatic insect abundance (Holt et al. 2014). Non-insect invertebrates in the same regulated system were more prevalent under low flow conditions than high flows (Holt et al. 2014). Filter-feeders and scrapers are negatively impacted by regulated flows, possibly because some net-spinning hydropsychids and other filter feeders may be easily damaged by abrasion and rapid flow changes below dams (Boon 1993, De Jalon et al. 1994). Scrapers may also decrease in relative abundance following impoundment in response to a decrease in macrophytes and periphyton biomass (De Jalon et al. 1994). These community-level assemblages are determined by species-specific adaptations and abilities to survive variable flows and temperatures.

Variable temperatures from dam releases in regulated rivers can also impact benthic invertebrates. During high reservoir periods, the water intake of the dam can occur below the thermocline and release water from much deeper depths than standard river flows, sending pulses of cold water into the river. Temperature changes like this can impact growth rates and emergence timing and duration of Ephemeroptera and Trichoptera, and the length of the time some Trichopteran species spend in late instars (Perry et al. 1986).

The dominance of Ephemeroptera and Chironomidae reported in the literature for regulated rivers is consistent with the benthic invertebrate community structure in the LCR, which is heavily influenced by regulated flows (Akers et al. 2021). Chironomidae were the dominant taxa by abundance in all sites in 2022, and Ephemeroptera were the third most common taxa. When assessed by biomass, Trichoptera, Ephemeroptera, and Chironomidae were the top three taxa in Norns Creek fan. These taxa are likely more successful and more prevalent because of their tolerance of regulated fluctuating flows and desiccation. Therefore, the mortality curve used in the productivity model is accurate for the LCR, as it is based on taxa with high desiccation tolerances.

Most studies that examine benthic invertebrate survival of dewatering measure mortality on a daily scale, or, more commonly, examine total population fluctuations over entire seasons. Lancaster and Ledger (2015) present proportional mortality curves and pre- and post-drying population densities for species of Chironomidae and Naididae from a drying experiment in artificial streams. Marchant and Hehir (1999) use population density, biomass, growth, and mortality to determine production for two Australian Trichoptera species. Another study determined survival and water loss rates for 12 Japanese species of Chironomid larvae (Suemoto et al. 2004). Poznańska et al. (2016) determined desiccation tolerances for a Chironomid species and an Oligochaete species, showing a strong difference in tolerance between the two, with the Chironomid showing far more tolerance to desiccation than the Oligochaete species. Examples of mortality curves can be found in the literature from different regions for the taxa of interest, but mortality curves determined from LCR processes and taxa are the most appropriate for the LCR productivity model.

1.1.1 Rainbow Trout Diets

Rainbow Trout diets can vary with food availability and can reflect the macroinvertebrate communities of their habitat (Fierro et al. 2016) but are often dominated by Ephemeroptera and Diptera (Meehan 1996, Whiting et al. 2014). Specifically, smaller Rainbow Trout consume smaller prey items like Ephemeroptera and Diptera in the drift and consume more large predatory benthic macroinvertebrates and smaller fish when trout grow to larger sizes above 150 mm (Whiting et al. 2014). However, diets can vary with region, stock, and season. A study of adult Rainbow Trout in the LCR showed that Diptera and Trichoptera made up greater than 50% of the benthic invertebrates in adult fish stomachs (Olson-Russello et al. 2019). The dominant taxa in RBT and Mountain Whitefish (*Prosopium williamsoni*) diets was Trichoptera, specifically net-spinning caddisflies (Hydropsychidae; Olson-Russello et al. 2019). Trichoptera and Diptera were the most abundant taxa available for fish consumption in the LCR during this study (Olson-Russello et al. 2019). While this study was a snapshot of fish diets during fall over two sampling years, it is likely that RBT foraged as generalists as their diets reflected the proportions of available taxa in the LCR at that time.

1.2. Conclusion

Mortality from desiccation will vary by season, species, and invertebrate life stage. Recolonization following dewatering and population reduction depends on species and the life stage during which invertebrates were dewatered. Both species resilience and recolonization combined influence post-drought biomass, and these metrics vary with season and accompanying physical processes, such as

temperature, light attenuation, saltation, and sand abrasion, among others. The species assemblage in the LCR is highly driven by regulated flows from the HLK dam, and in turn determines the diets of Rainbow Trout.

Physical habitat characteristics as velocity, light attenuation, temperature, and substrate size influence invertebrate community structure and productivity. The growth and mortality curves used in the model are based on peak biomass, however, abrasion from sediment and variable flows can remove periphyton and benthic invertebrates from rocks and result in biomass loss (Schleppe et al. 2013). Light attenuation is a major determinant in macrophyte and periphyton productivity, which in turn influences population densities of scraper invertebrates (de Jalon et al. 1994). The objective of the Norns Creek fan productivity model is to gain a relative understanding of the effect of flows on overall productivity. This question has a broad scope, and the model does not address physical characteristics beyond inundation. Including physical characteristics on a microsite scale may improve model precision but would require much larger and more complex datasets than are already included.

While physical site characteristics may be costly and labor intensive, additional flow metrics may be included to improve the accuracy of the hydraulic model, and the resulting benthic productivity accrual estimates. For example, recent modeling of the Columbia River's benthic productivity for the Columbia River Treaty used a suite of flow parameters as predictors, rather than water level alone (Akers et al. 2021). These predictors were variability of daily flows, number of high flow events, mean duration of high flow events, fall rate of decreasing flows, daily flow magnitude changes, and maximum annual flow (Akers et al. 2021). Including these flow parameters may help detect the specific impacts of flow variability on benthic productivity, rather than a comparison between the highly variable ON and OFF years alone.

The mortality curve used to predict invertebrate death rates in the model was adapted from previous studies conducted for the Columbia Power Corporation, and was based on Simuliidae, and Hydropsychidae, two of the most abundant LCR invertebrate taxa (Schleppe et al. 2013; Schleppe et al. 2015). RBT in the LCR can forage as generalists, and the dominant taxa in the diets of adults and some juvenile taxa was Trichoptera. Therefore, the mortality curve used in the model is specific to LCR taxa that constitute a large portion of RBT diets. Furthermore, the mortality curves were derived from data collected in the LCR, so they reflect the specific site conditions of the river system. We conclude that the effort and associated cost of producing and integrating mortality curves for more taxa would not significantly increase model precision.

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APPENDIX B LIGHT ATTENUATION MODELLING

Lower Columbia River Light Attenuation and Reflectance 2022

Thorley, J.L. & Kortello, A.

Draft: 2022-12-05 12:18:00

The suggested citation for this analytic appendix is:

Thorley, J.L. & Kortello, A. (2022) Lower Columbia River Light Attenuation and Reflectance 2022. A Poisson Consulting Analysis Appendix. URL: <https://www.poissonconsulting.ca/f/130474005>.

Background

The primary goal of the analysis is to address the following question:

How does light in the Lower Columbia River attenuate with depth?

What is the light surface reflectance in the Lower Columbia River?

Methods

Data Collection

In order to estimate light attenuation, Ecoscape recorded light (PAR) and turbidity levels at different depths at various sites. Ecoscape also recorded light levels at the surface and 0.01 m below the surface in order to estimate surface reflectance.

The data were provided as Excel spreadsheets.

Data Preparation

The data were prepared for analysis using R version 4.2.2 (R Core Team 2022).

Statistical Analysis

Model parameters were estimated using Bayesian methods. The estimates were produced using JAGS (Plummer 2003). For additional information on Bayesian estimation the reader is referred to McElreath (2020).

Unless stated otherwise, the Bayesian analyses used weakly informative normal and half-normal prior distributions (Gelman, Simpson, and Betancourt 2017). The posterior distributions were estimated from 1500 Markov Chain Monte Carlo (MCMC) samples thinned from the second halves of 3 chains (Kery and Schaub 2011, 38–40). Model convergence was confirmed by ensuring that the potential scale reduction factor $\hat{R} \leq 1.05$ (Kery and Schaub 2011, 40) and the effective sample size (Brooks et al. 2011) $ESS \geq 150$ for each of the monitored parameters (Kery and Schaub 2011, 61).

The parameters are summarised in terms of the point *estimate*, *lower* and *upper* 95% compatibility limits (Rafi and Greenland 2020) and the surprisal *s-value* (Greenland 2019). The estimate is the median (50th percentile) of the MCMC samples while the 95% CLs are the 2.5th and 97.5th percentiles. The *s-value* indicates how surprising it would be to discover that the true value of the parameter is in the opposite direction to the estimate (Greenland 2019). An *s-value* of > 4.3 bits, which is equivalent to a significant *p-value* < 0.05 (Kery and Schaub 2011; Greenland and Poole 2013), indicates that the surprise would be equivalent to throwing at least 4.3 heads in a row.

The condition that parameters describing the effects of secondary (nuisance) explanatory variable(s) have significant *p-values* was used as a model selection heuristic (Kery and Schaub 2011). Based on a similar argument, the condition that random effects have a standard deviation with a lower 95% compatibility interval (CL) $> 5\%$ of the estimate was used as an additional model selection heuristic.

Model adequacy was assessed via posterior predictive checks (Kery and Schaub 2011). More specifically, the first four central moments (mean, variance, skewness and kurtosis) for the deviance residuals were compared to the expected values by simulating new residuals. In this context the *s-value* indicates how surprising each observed metric is given the estimated posterior probability distribution for the residual variation.

Where computationally practical, the sensitivity of the parameters to the choice of prior distributions was evaluated by increasing the standard deviations of all normal, half-normal and log-normal priors by an order of magnitude and then using \hat{R} to evaluate whether the samples were drawn from the same posterior distribution (Thorley and Andrusak 2017).

The results are displayed graphically by plotting the modeled relationships between individual variables and the response with the remaining variables held constant. In general, continuous and discrete fixed variables are held constant at their mean and first level values, respectively, while random variables are held constant at their average values (expected values of the underlying hyperdistributions) (Kery and Schaub 2011, 77–82).

The analyses were implemented using R version 4.2.2 (R Core Team 2022) and the *mbr* family of packages.

Model Descriptions

Light Attenuation

The following equation describes how light attenuates in water

$$E_d = E_0 \cdot \exp(-K_d \cdot y)$$

where E_0 is the initial irradiance, E_d is the irradiance at distance y and K_d is the diffuse attenuation coefficient (Julian, Doyle, and Stanley 2008).

Key assumptions of the surface reflectance model include:

- There are no measurement errors in E_0 .

- The residual variation in E_d is log-normally distribution.

As all the turbidity readings were < 2 NTU, turbidity was not included in the model.

Euphotic Zone

Ignoring surface reflectance, the euphotic zone (depth at which light is 1% of the surface) is $-\log(0.01)/K_d$.

Surface Reflectance

The relationship between the irradiance at the surface (E_s) and the irradiance 0.01 m below the surface (E_0) was modelled using the relationship

$$E_0 = E_s \cdot r \cdot \exp(-K_d \cdot 0.01)$$

where r is the reflection coefficient (Julian, Doyle, and Stanley 2008) and K_d was the estimate from the attenuation model.

Key assumptions of the surface reflectance model include:

- There are no measurement errors in E_s .
- The residual variation in E_0 is log-normally distributed.

As all the turbidity readings were < 2 NTU, turbidity was not included in the model.

Euphotic Zone

Including surface reflectance, the euphotic zone (depth at which light is 1% of the surface) is $-\log(0.01/r)/K_d$.

Model Templates

Attenuation

```
.model{
  Kd ~ dnorm(-1, 2^-2)
  sLight2 ~ dnorm(0, 1^-2) T(0, )

  for (i in 1:length(Light2)) {
    eKd[i] <- exp(Kd)
    eLight2[i] <- Light[i] * exp(-eKd[i] * Distance[i])

    Light2[i] ~ dlnorm(log(eLight2[i]), sLight2^-2)
  }
}
```

Block 1. Model description.

Reflectance

```
.model{
  rho ~ dunif(0, 1)
  sLight2 ~ dnorm(0, 1^-2) T(0,)
```

```

Kd ~ dnorm(-1.1, 0.07^-2) T(-1.2, -1)

for (i in 1:length(Light2)) {
  eKd[i] <- exp(Kd)
  eLight2[i] <- Light[i] * exp(-eKd[i] * 0.01) * rho
  Light2[i] ~ dlnorm(log(eLight2[i]), sLight2^-2)
}

```

Block 2. Model description.

Results

Tables

Attenuation

Table 1. Parameter descriptions.

Parameter	Description
Distance	The distance (y) in m
eKd	The expected diffuse attenuation coefficient in m^{-1}
eLight2	Expected Light2
Kd	The intercept for $\log(eKd)$
Light	The initial irradiance (E_0) in lx
Light2	The irradiance at distance (E_d) in lx
sLight2	SD of measurement error in Light2

Table 2. Model coefficients.

term	estimate	lower	upper	svalue
Kd	-1.0893895	-1.2337906	-0.9652501	10.55171
sLight2	0.1360966	0.1223378	0.1520452	10.55171

Table 3. Model convergence.

n	K	nchains	niters	nthin	ess	rhat	converged
158	2	3	500	20	1448	1	TRUE

Table 4. Euphotic zone depth (m) ignoring surface reflectance (with 95% CIs).

estimate	lower	upper
13.68868	15.81518	12.09062

Table 5. Model posterior predictive checks.

moment	observed	median	lower	upper	svalue
--------	----------	--------	-------	-------	--------

moment	observed	median	lower	upper	svalue
zeros	NA	NA	NA	NA	NA
mean	-0.0075977	-0.0043807	-0.1453396	0.1471690	0.0488765
variance	0.9903349	1.0024161	0.7928849	1.2275792	0.1098016
skewness	-0.8537086	0.0032855	-0.3773260	0.3852552	10.5517083
kurtosis	1.4197764	-0.0907401	-0.5876605	0.9413734	6.3037807

Table 6. Model sensitivity.

n	K	nchains	niters	rhat_1	rhat_2	rhat_all	converged
158	2	3	500	1.001	1.001	1	TRUE

Reflectance

Table 7. Parameter descriptions.

Parameter	Description
Light	The irradiance at the surface (E_s) in lx
Light2	The irradiance just below the surface (E_0) in lx
rho	Reflection coefficient (r)
sLight2	SD of measurement error in Light2

Table 8. Model coefficients.

term	estimate	lower	upper	svalue
Kd	-1.0988725	-1.1907383	-1.0087445	10.55171
rho	0.5927078	0.5613640	0.6235446	10.55171
sLight2	0.1894281	0.1596365	0.2347296	10.55171

Table 9. Model convergence.

n	K	nchains	niters	nthin	ess	rhat	converged
56	3	3	500	1	806	1.004	TRUE

Table 10. Euphotic zone depth (m) including surface reflectance (with 95% CIs).

estimate	lower	upper
12.13392	14.19308	10.57472

Table 11. Model posterior predictive checks.

moment	observed	median	lower	upper	svalue
zeros	NA	NA	NA	NA	NA
mean	-0.0082173	0.0013017	-0.2604660	0.2525078	0.0709181

moment	observed	median	lower	upper	svalue
variance	0.9828269	0.9963357	0.6539335	1.3828684	0.0830842
skewness	-0.9400043	0.0032479	-0.6463506	0.6087133	8.9667458
kurtosis	0.2297131	-0.2043870	-0.8882035	1.2440925	1.0943274

Table 12. Model sensitivity.

n	K	nchains	niters	rhat_1	rhat_2	rhat_all	converged
56	3	3	500	1.004	1.003	1.002	TRUE

Figures

Attenuation

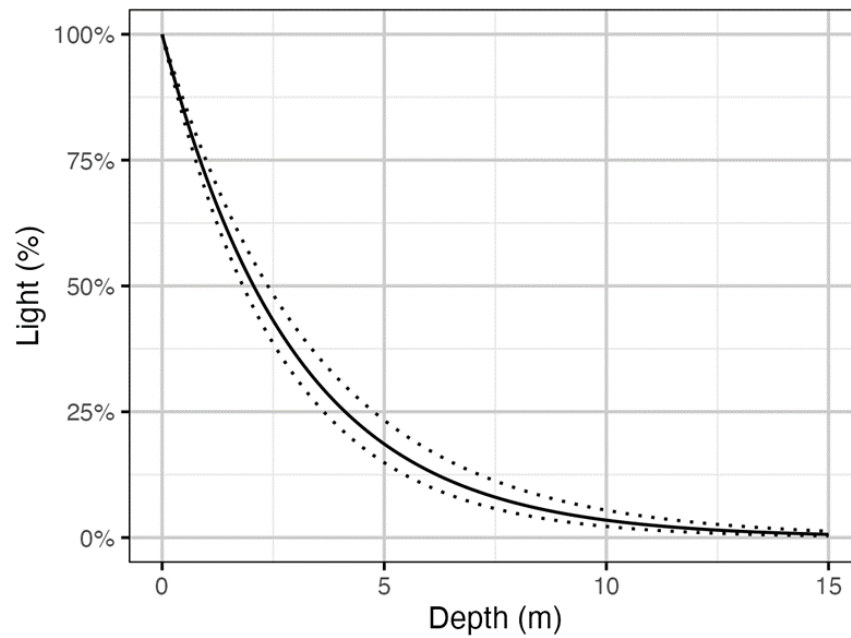


Figure 1. Light attenuation by depth.

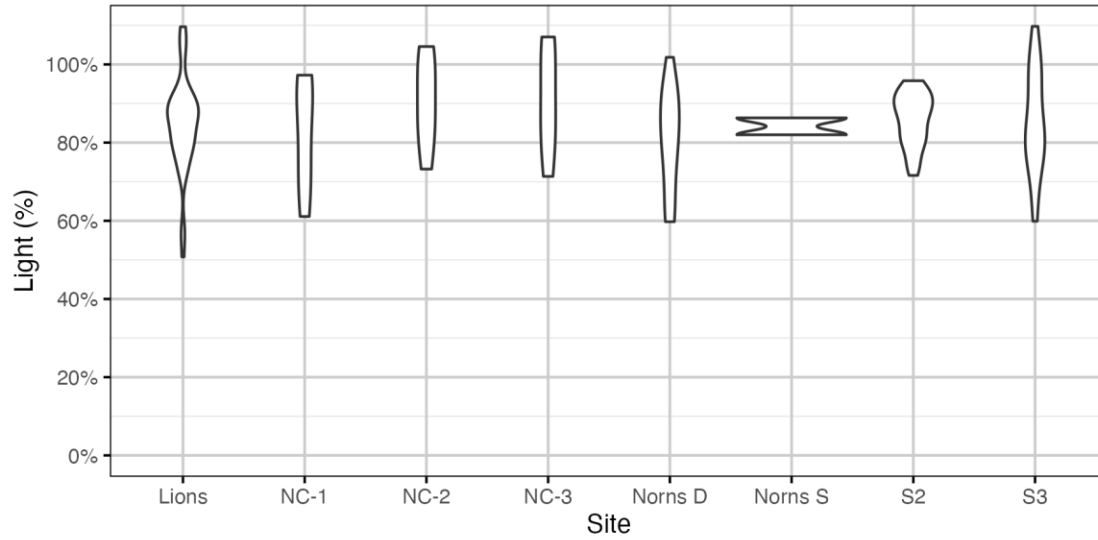


Figure 2. Light attenuation at a depth change of 0.5 m by site.

Reflectance

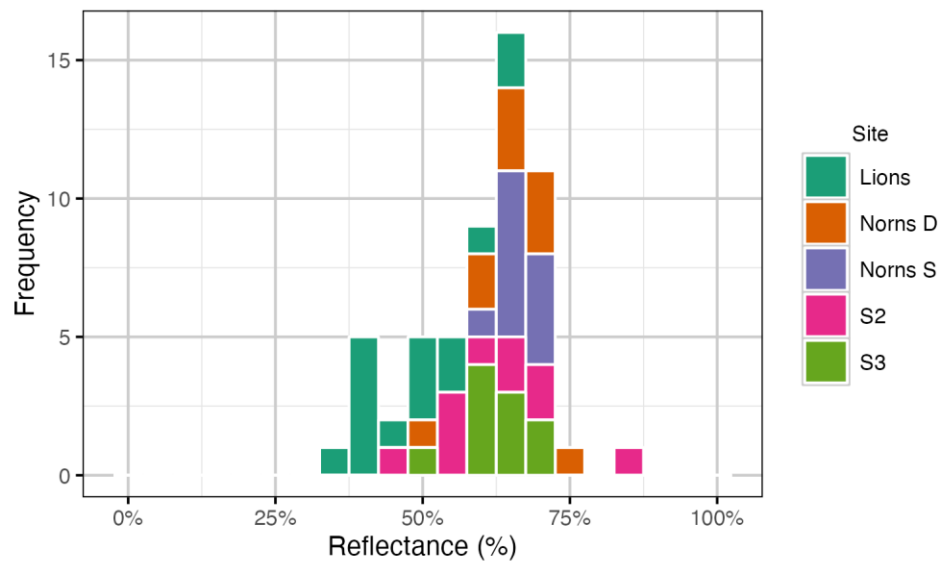


Figure 3. Surface reflectance by site.

Acknowledgements

The organisations and individuals whose contributions have made this analytic appendix possible include:

- BC Hydro
- Ecoscape
 - Rachel Plewes
 - Mary Ann Olson-Russello

– Jason Schleppe

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APPENDIX C MEASURED PERIPHYTON SUMMARY

Appendix C Data to support Norns Creek Fan Periphyton Section 4.3.1

Table A-1: Summary of Norns Creek fan periphyton productivity compared to the LCR R2 reach.

Variable	Reach	Mean	St. Dev.	Median	Min	Max	n
Abundance	NC	3.34e+04	8.36e+04	9.09e+03	2.27e+03	9.20e+05	15
	R2	3.57e+04	9.12e+04	9.09e+03	2.27e+03	7.57e+05	5
Biovolume	NC	0.12	0.70	0.02	0.00	15.96	15
	R2	0.08	0.18	0.02	0.00	1.60	5

Table A-2: Norns Creek fan periphyton productivity by site compared to the control R2-S1 site.

Site	Total Abundance	Species Richness	Simpson Index	Shannon Index	Shannon Equitability	Effective Species	Total Biovolume
NC-1	7.65e+06	69	0.96	3.99	0.94	53.89	24.13
NC-2	7.15e+06	85	0.97	4.26	0.96	70.65	31.88
NC-3	5.92e+06	87	0.97	4.13	0.92	62.20	20.04
R2-S1	5.96e+06	69	0.96	3.87	0.91	47.93	13.89

Table A-3: Periphyton productivity and diversity metrics by transect depth from 2023.

Site	Transect	Total Abundance	Species Richness	Simpson Index	Shannon Index	Shannon Equitability	Effective Species	Total Biovolume
NC-1	S	9.25e+05	39	0.84	2.56	0.70	12.88	1.87
	MS	2.67e+06	36	0.83	2.36	0.66	10.64	7.36
	M	1.27e+06	40	0.85	2.58	0.70	13.23	5.17
	MD	7.54e+05	37	0.87	2.67	0.74	14.49	2.91
	D	2.03e+06	37	0.86	2.48	0.69	11.97	6.82
NC-2	S	1.20e+06	38	0.82	2.46	0.68	11.67	2.66
	MS	1.32e+06	45	0.89	2.73	0.72	15.30	2.45
	M	1.37e+06	53	0.86	2.90	0.73	18.13	4.69
	MD	1.20e+06	39	0.90	2.83	0.77	16.88	2.78
	D	2.06e+06	53	0.84	2.52	0.63	12.40	19.31
NC-3	S	7.79e+05	29	0.79	2.17	0.64	8.76	1.55
	MS	1.88e+06	52	0.82	2.47	0.62	11.79	6.05
	M	1.36e+06	39	0.87	2.63	0.72	13.81	5.95
	MD	1.32e+06	49	0.90	2.96	0.76	19.26	5.26
	D	5.89e+05	34	0.88	2.71	0.77	14.99	1.23
R2-S1	S	1.10e+06	45	0.88	2.83	0.74	16.88	2.80
	MS	1.35e+06	34	0.77	2.12	0.60	8.29	1.93
	M	1.13e+06	39	0.81	2.48	0.68	11.91	4.64
	MD	6.70e+05	29	0.87	2.50	0.74	12.13	1.74
	D	1.71e+06	20	0.76	1.92	0.64	6.84	2.79

Table A-4: Dominant periphyton taxa ranked by abundance.

Reach-Site				
Dominant Taxa Rank	NC-1	NC-2	NC-3	R2-S1
1	<i>Achnanthydium minutissimum</i>	<i>Achnanthydium minutissimum</i>	<i>Achnanthydium minutissimum</i>	<i>Achnanthydium minutissimum</i>
2	<i>Rossithidium anastasiae</i>	<i>Rossithidium anastasiae</i>	<i>Rossithidium anastasiae</i>	<i>Rossithidium anastasiae</i>
3	<i>Achnanthydium microcephalum</i>	<i>Anacystis cyanea</i>	<i>Cymbella minuta</i>	<i>Achnanthydium microcephalum</i>
4	<i>Aphanocapsa</i> sp.	<i>Achnanthydium microcephalum</i>	<i>Aphanocapsa</i> sp.	<i>Achnanthydium deflexum</i>
5	<i>Achnanthydium deflexum</i>	<i>Fragilaria vaucheriae</i>	<i>Achnanthydium microcephalum</i>	picoflagellates
6	<i>Gomphonella olivacea</i>	<i>Achnanthydium deflexum</i>	<i>Fragilaria vaucheriae</i>	<i>Gomphonella olivacea</i>
7	<i>Anacystis cyanea</i>	<i>Cymbella minuta</i>	<i>Fragilaria</i> cf. <i>rumpens</i>	<i>Fragilaria vaucheriae</i>
8	picoflagellates	<i>Gomphonella olivacea</i>	<i>Achnanthydium deflexum</i>	<i>Encyonema minutum</i>
9	<i>Cyclotella atomus</i>	picoflagellates	<i>Tabellaria fenestrata</i>	microflagellates
10	<i>Fragilaria vaucheriae</i>	<i>Aphanocapsa</i> sp.	<i>Planktothrix</i> sp.	<i>Planktolyngbya limnetica</i>

Table A-5: Dominant periphyton taxa ranked by biovolume.

Reach-Site				
Dominant Taxa Rank	NC-1	NC-2	NC-3	R2-S1
1	<i>Didymosphenia geminata</i>	<i>Didymosphenia geminata</i>	<i>Tabellaria flocculosa</i>	<i>Rossithidium anastasiae</i>
2	<i>Ulnaria ulna</i>	<i>Rossithidium anastasiae</i>	<i>Didymosphenia geminata</i>	<i>Achnanthydium minutissimum</i>
3	<i>Rossithidium anastasiae</i>	<i>Achnanthydium minutissimum</i>	<i>Rossithidium anastasiae</i>	<i>Didymosphenia geminata</i>
4	<i>Achnanthydium minutissimum</i>	<i>Ulnaria ulna</i>	<i>Achnanthydium minutissimum</i>	<i>Melosira varians</i>
5	<i>Tribonema vulgare</i>	<i>Gomphonella olivacea</i>	<i>Ulnaria ulna</i>	<i>Gomphoneis herculeana</i>
6	<i>Gomphonella olivacea</i>	<i>Cocconeis pediculus</i>	<i>Fragilaria</i> cf. <i>rumpens</i>	<i>Ulnaria ulna</i>
7	<i>Navicula cryptotenella</i>	<i>Ulnaria acus</i>	<i>Cymbella minuta</i>	<i>Gomphonella olivacea</i>
8	<i>Ulnaria acus</i>	<i>Fragilaria</i> cf. <i>rumpens</i>	<i>Tabellaria fenestrata</i>	<i>Ulnaria acus</i>
9	<i>Cymbella parva</i>	<i>Melosira varians</i>	<i>Melosira varians</i>	<i>Navicula cryptotenella</i>
10	<i>Achnanthydium deflexum</i>	<i>Fragilaria vaucheriae</i>	<i>Ulnaria acus</i>	<i>Bodo</i> sp.

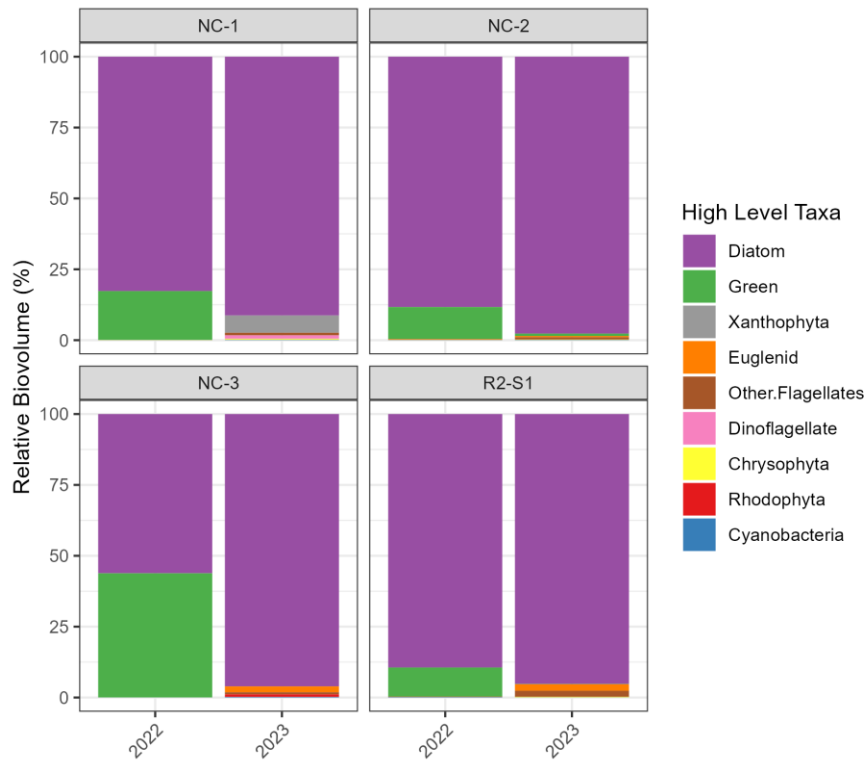
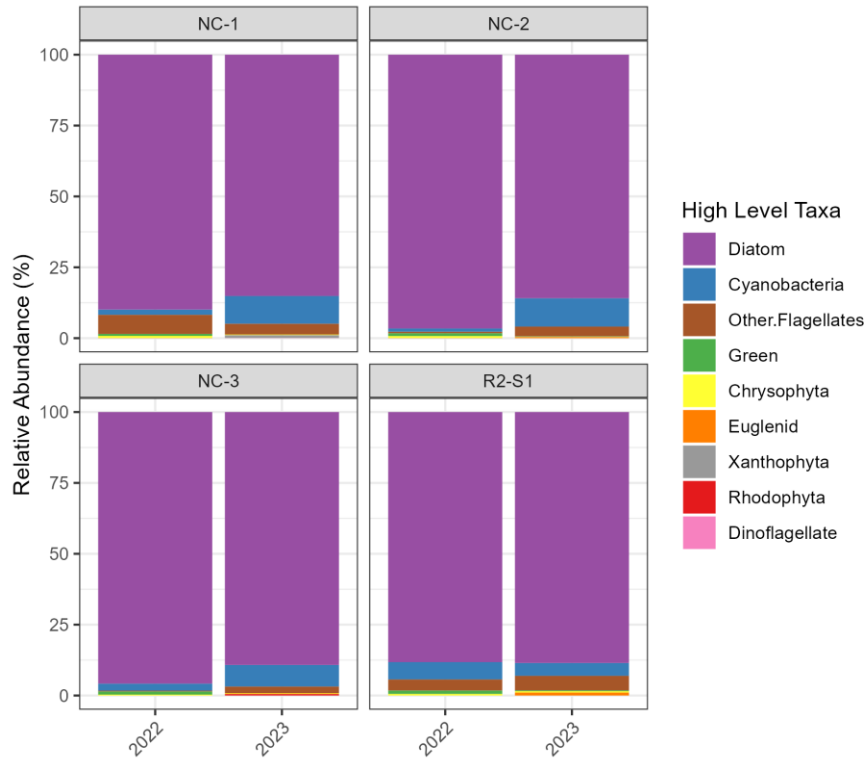


Figure A-1: Abundance and biovolume for the major periphyton taxonomic classes for Norns Creek fan, 2023 data.

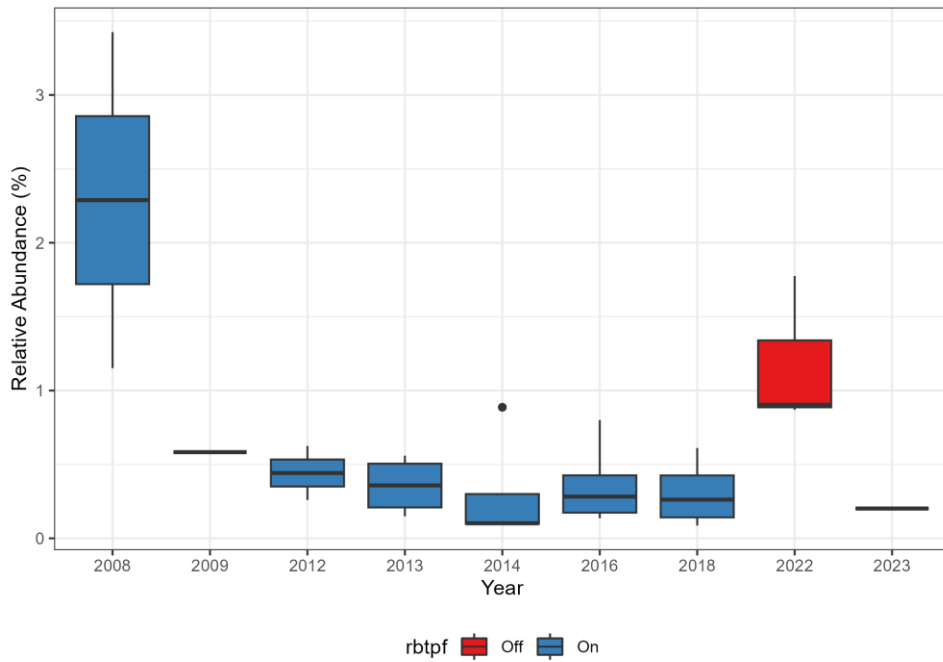
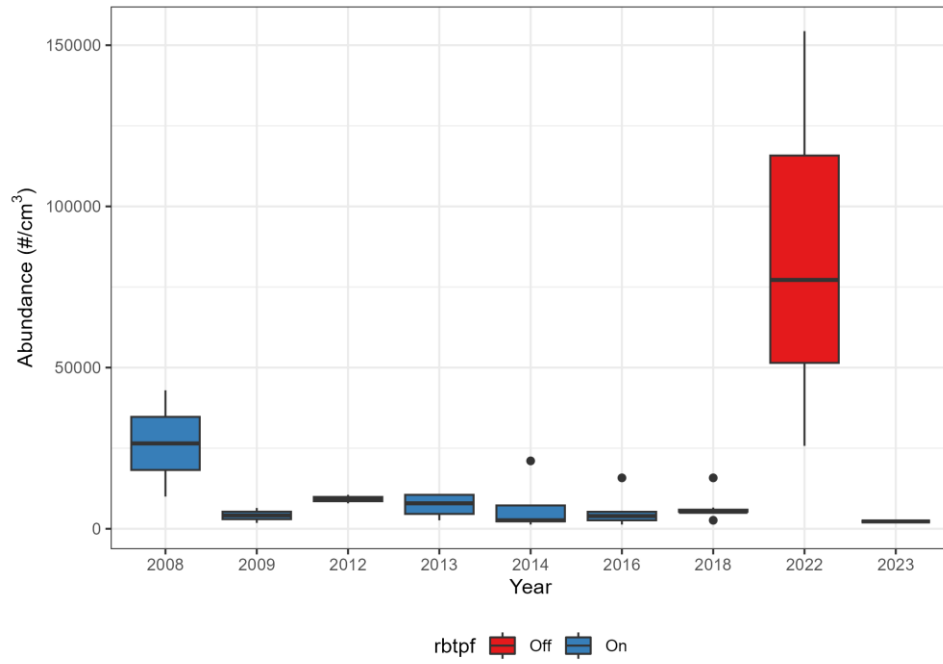


Figure A-2: Observed Didymo abundance in ON and OFF years during the study period.