BC hydro

Aberfeldie Water Use Plan

ABERFELDIE WATER USE PLAN: PRIMARY AND SECONDARY PRODUCTIVITY MONITORING

Reference: ABFMON#2

Implementation Years 3 - 4

Study Period: 2009 to 2012

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May 31, 2013

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LIMNOTEK

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Submitted to

BC Hydro
Burnaby, B.C.
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Cover photo: New Aberfeldie power generating station on the Bull River, 2012. Photo by Chris Perrin.

EXECUTIVE SUMMARY

A before-after-control-impact assessment was used to examine the effect of a 2009 upgrade to the Aberfeldie hydropower facilities on benthic invertebrate and periphyton assemblages in the Bull River, British Columbia. Before the upgrade, water flowed over the dam spillway during eight to nine months from late March to late December but spill is now limited to three to four months from late April to late July. During the new non-spill periods in March through April and August through December, minimum flow is released from the dam according to recommendations from the Water Use Planning Consultative Committee and Fisheries Technical Committee (BC Hydro 2006). Actual timing of spill varies according to river flow and capacity of the turbines to pass that flow. During the non-spill periods before the upgrade (January – February), flow from the dam was limited to leakage that was estimated to be 0.05 $m^3 \cdot s^{-1}$ but minimum flow that is now released to the river in that same period is 0.25 $m^3 \cdot s^{-1}$. Minimum flow is 0.5 $m^3 \cdot s^{-1}$ in April through May, 2.0 $m^3 \cdot s^{-1}$ in June through September, 0.5 $m^3 \cdot s^{-1}$ in October through, and 0.25 $m^3 \cdot s^{-1}$ in December through March. In the biologically productive period of August and September, median flow in the diversion reach declined from 6 $m^3 \cdot s^{-1}$ before the upgrade to 2 $m^3 \cdot s^{-1}$ afterwards. This study examined effects of this change in flow on benthic invertebrate and periphyton metrics in the diversion reach and on the same metrics in the downstream reach during August through September. The biological samples were collected from a control reach upstream of the headpond, a riffle segment within the diversion reach between the dam and powerhouse, and a reach downstream of the powerhouse. Samples were collected from each location during each of two years before the upgrade (2005 and 2006) and three years afterwards (2009, 2010, 2012). Measurements of habitat attributes were completed at the time of biological sampling.

The combined density of mayflies, stoneflies, and caddisflies (commonly known as the EPT, an acronym for Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)) that are considered most sensitive to environmental change among aquatic invertebrates increased by 50% in the diversion reach due to the reduced flow. This change was statistically significant. The response of EPT was hypothesized to be caused by lower water velocities and thus lower shear compared to conditions before the upgrade. The EPT density did not change in relation to change of operations in the downstream reach. Chironomids, other non-EPT invertebrates, diversity of invertebrates, and periphyton biomass were not affected either by the change in flow in the diversion reach or change in operations in the downstream reach. The statistical tests showing no effect of change in flow and operations on the non-EPT assemblages had low power due to few years of monitoring, which means that the tests had little chance of detecting a flow or operations effect on the non-EPT metrics if it was present. Given that the EPT are known to be important food organisms for fish that use

riffle habitat in the diversion reach, the findings showed that lower flow in late summer increased the density of fish food organisms in continuously wetted areas of the diversion reach.

With completion of this report, management questions and hypotheses associated with ABFMON2 have been addressed as noted in the following table.

Status of objectives, management questions, and hypotheses for ABFMON#2

Project number	Objectives	Management questions	Management hypotheses	Status following work in 2012	Page(s) showing the result for ABFMON2
ABFMON#2	To quantify changes in productivity in the diversion reach of the Bull River due to the operating regime associated with the redeveloped Aberfeldie facility	What is the net effect of the post redevelopment flow regime on the community composition, diversity, abundance, and peak biomass of periphyton in the diversion reach of Bull River?	Ho1: The implementation of the post upgrade 2 $m^3 \cdot s^{-1}$ minimum summer flow release does not change the peak biomass of periphyton in the diversion reach of the Bull River from pre-upgrade conditions.	Ho1 is accepted.	38-40, 43-45
			Ho2: The implementation of the post upgrade 2 $m^3 \cdot s^{-1}$ minimum summer flow release does not change the diversity of periphyton in the diversion reach of the Bull River from pre-upgrade conditions.	Ho2 is accepted	37-38, 42-43
		What is the net effect of the post redevelopment flow regime on the community composition, diversity, and abundance of benthic invertebrates in the diversion reach of Bull River?	Ho3: The implementation of the post upgrade 2 $m^3 \cdot s^{-1}$ minimum summer flow release does not change the total abundance, biomass and diversity of benthic invertebrates in the diversion reach of the Bull River from pre-upgrade conditions.	Ho3 is accepted for total invertebrates but density of the assemblage of mayflies, stoneflies, and caddisflies significantly increased with implementation of minimum flow in the diversion reach.	28-36, 40-42

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1 INTRODUCTION

Aberfeldie is a run-of-the-river power generation project that was built on the Bull River, approximately 35 km east of Cranbrook, British Columbia in 1922 (http://www.bchydro.com/info) (Figures 1 and 2). Works include the Aberfeldie Dam, a headpond upstream of the dam, a penstock that conveys water from the headpond to a power generating station downstream from which water is discharged back into the Bull River. In June 2009, BC Hydro completed an upgrade to Aberfeldie that increased flows from 9.9 $m^3 \cdot s^{-1}$ through the original powerhouse to a maximum of 40 $m^3 \cdot s^{-1}$ through a new powerhouse (BC Hydro 2009). Average annual energy production increased from 5 MW produced from one Francis[1] turbine to 25 MW produced from three Francis turbines. Before the upgrade, water flowed over the dam spillway during eight to nine months from late March to late December but spill is now limited to three to four months from late April to late July. During the new non-spill periods in March through April and August through December, minimum flow is released from the dam according to recommendations from the Water Use Planning Consultative Committee and Fisheries Technical Committee (BC Hydro 2006). Actual timing of spill varies according to river flow and capacity of the turbines to pass that flow. During the non-spill periods before the upgrade (January – February), flow from the dam was limited to leakage that was estimated to be 0.05 $m^3 \cdot s^{-1}$ but minimum flow that is now released to the river in that same period is 0.25 $m^3 \cdot s^{-1}$. Minimum flow is 0.5 $m^3 \cdot s^{-1}$ in April through May to support fish movement, 2.0 $m^3 \cdot s^{-1}$ in June through September to support fish movement and benthic invertebrate production, 0.5 $m^3 \cdot s^{-1}$ in October through November to support fish movement, and 0.25 $m^3 \cdot s^{-1}$ in December through March to maintain winter habitat for fish. In the biologically productive period of August and September, median flow in the diversion reach declined from 6 $m^3 \cdot s^{-1}$ before the upgrade to 2 $m^3 \cdot s^{-1}$ afterwards.

The diversion reach consists of a canyon segment and test segment (Figures 3 and 4). The 840 m long upstream "canyon segment" has moderate to high gradient, narrow steep walls, and substrate consisting of bedrock and large boulder. At a bank-full flow of 120 $m^3 \cdot s^{-1}$, habitat includes deep bedrock confined pools (57% of area), shallower pools (5%), cascades (22%), small riffles (9%), and step-pools (8%) (Cope 2005). There are several barriers in the canyon that are impassable to fishes (Figure 4), with the first upstream barrier being located near the canyon outflow. We hypothesize that the deep water, steep gradient, high water velocities, and absence of gravel and cobble limits benthos production in this canyon segment. Downstream of the canyon is a 335 m segment between the first upstream barrier and the powerhouse tailrace hereafter referred to as the "test segment" (Figures 3 and 4). It has lower gradient and is less confined than the canyon. Habitat consists of a deep bedrock pool approximately 30 m in length, located at the canyon outlet, and the remainder is riffle extending downstream

[1] A Francis turbine is a common type of water turbine that was developed by James B. Francis in Lowell, Massachusetts in 1848 (http://en.wikipedia.org/wiki/Francis_turbine#Development).

to a backwater pool formed behind the powerplant tailrace. Substrate in the test segment includes bedrock, boulder, cobble, gravel, and small amounts of sand. Fishes in the diversion reach can potentially ingest invertebrates produced in the test segment and larval insects that drift from the shallow headpond upstream of the Aberfeldie Dam.

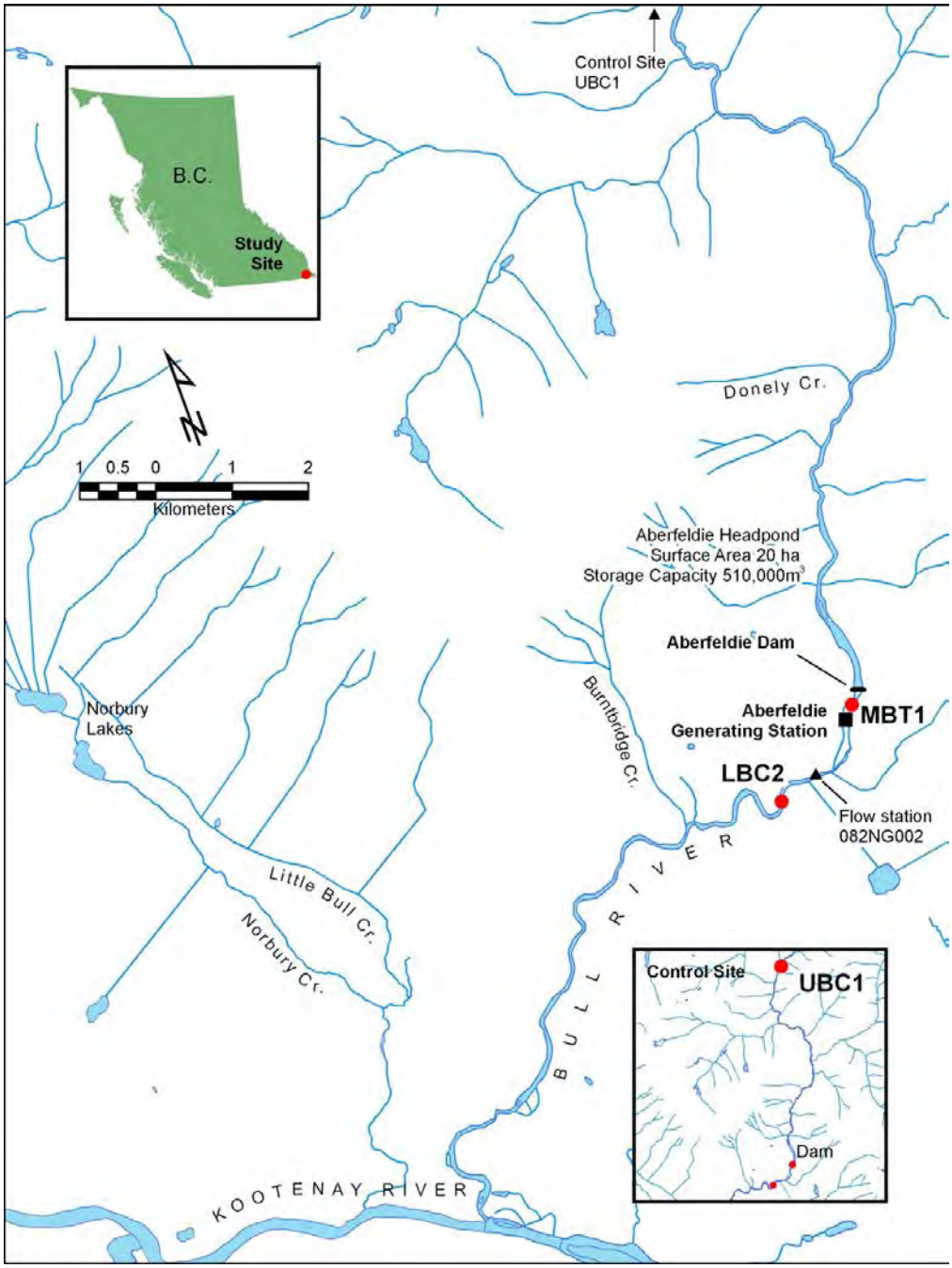


Figure 1 Bull River study area showing the geographic location and layout of the Aberfeldie generating facilities and sampling sites. UBC1 = Upper Bull Control site, MBT1 = Middle Bull Treatment site, LBC2 = Lower Bull Control site.

Figure 2. Original Aberfeldie powerhouse dating to 1922 (top, photo taken in 2006) and the upgraded powerhouse at the same location in 2012 (bottom).

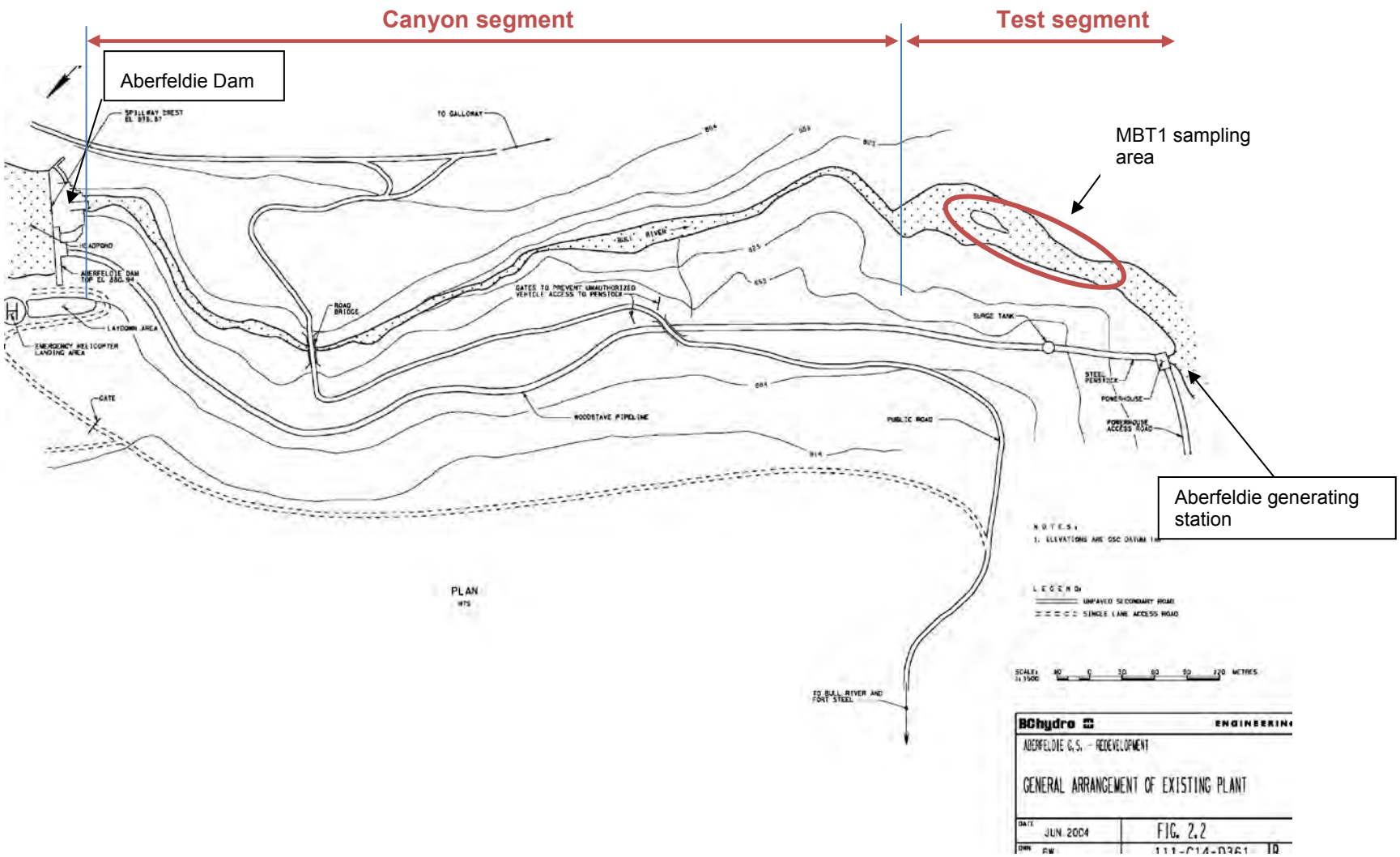
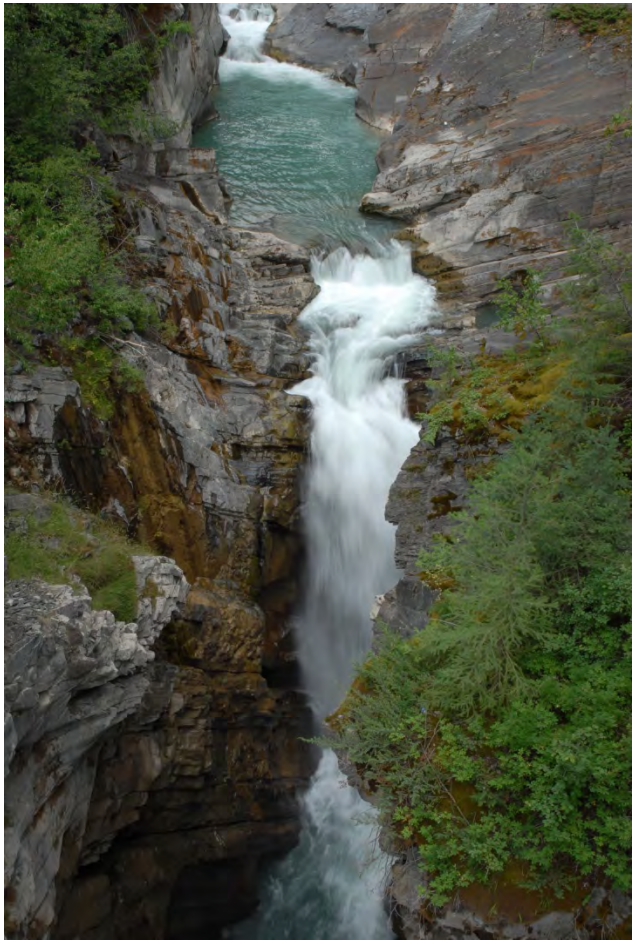


Figure 3. Map of diversion reach of the Bull River showing the canyon and test segments and the MBT1 sampling site. Map modified from BC Hydro (2004).



Canyon segment of the diversion reach

Test segment of the diversion reach

Figure 4. Image of the canyon segment (left) and test segment (right) of the diversion reach at minimum flow of 2 $m^3 \cdot s^{-1}$.

At the prescribed minimum flow of 2 $m^3 \cdot s^{-1}$ during summer, Cope (2005) estimated that 65% of wetted habitat area would be lost in August and 52% would be lost in September compared to habitat areas at the pre-upgrade flows. Lost habitat may result in lower total amounts of periphyton (algae with an assemblage of bacteria and fungi) and benthic invertebrates that together form the basis of the food chain for stream-dwelling fishes.

To compensate for lost habitat, 5,290 m^2 of side channel habitat was constructed in an area approximately 500 m downstream of the Aberfeldie generating station (McPherson et al. 2010). The side channel area is greater than the 3,600 m^2 of habitat that Cope (2005) estimated would be lost from the diversion reach if flows declined from the average monthly flows before the Aberfeldie upgrade to minimum flow after the upgrade. The side channel was designed with optimum substrate particle sizes to support benthic assemblages and provide complex spawning and rearing habitat for fishes. Hence, its productive capacity per unit area should be greater than that of the diversion reach, given adequate flow. This added capacity means that cumulative availability of fish food organisms in the diversion reach and side channel are expected to at least equal or exceed amounts that were present in the diversion reach before the upgrade.

The consultative committee proposed four management questions to address uncertainty about the effect of change in flow in the diversion reach on benthic assemblages and uncertainty about benefits of the side channel in compensating for possible loss of productive capacity in the diversion reach caused by the Aberfeldie upgrade (BC Hydro 2008). Those questions are as follows:

Management question 1: What is the net effect of the post redevelopment flow regime on the community composition, diversity, abundance, and peak biomass of periphyton in the diversion reach of Bull River?

Management question 2: What is the net effect of the post redevelopment flow regime on the community composition, diversity, biomass and abundance of benthic invertebrates in the diversion reach of Bull River?

Management question 3: If changes in the benthic community associated with post-redevelopment facility operations are detected, does the prescribed flow regime, combined with the productive capacity realized from the compensation habitat achieve the Aberfeldie Redevelopment project compensation goal of no-net-loss of productive capacity?

Management question 4: Is there an alternate minimum instream flow discharge that, in combination with the productive capacity realized from the compensation habitat, achieves the Aberfeldie Redevelopment project compensation goal of no-net-loss of productive capacity in the diversion reach of the Bull River?

This report addresses management questions 1 and 2 while 3 and 4 are answered in a companion report (Perrin and Bennett, 2013b). In this study, "productive capacity" is defined by several metrics. For periphyton it includes accrual of biomass to reach a peak amount, defined as peak biomass (PB), during the incubation of growth media in the river for a defined period of time. It also includes counts and biovolume of algal cells, by species or other taxonomic level. For benthic invertebrates, productive capacity is defined by counts of individual animals that can be used to derive a variety of metrics suitable for testing the effects of change in flow in the diversion reach on invertebrate assemblages. Based on these definitions biological "production" in its true sense was not measured but capacity of the river channel to support benthic assemblages was measured. The phrase "productive capacity" is a convenient term applied to these measurements, and it consistent with the accepted definition of productive capacity for fish as defined by Fisheries and Oceans Canada (maximum natural capability of habitats to produce healthy fish, safe for human consumption, or to support or produce aquatic organisms upon which fish depend; DFO 2013).

Two processes can occur either together or independently to modify benthic assemblages in the diversion reach between years before and after the Aberfeldie upgrade. One is a potential effect of flow on density and biomass of biological assemblages within continuously wetted areas, which is addressed in this report; and the other is change in wetted area hosting benthic communities, which is addressed in a companion report (Perrin and Canning 2010). The net effect of these two processes on biological assemblages in the diversion reach due to the Aberfeldie upgrade and compensation for the potential change by biological production in the side channel is addressed by Perrin and Bennett (2013b).

2 METHODS

2.1 Study Scope

The focus of analyses was on benthic invertebrates because they are the assemblage that provides food for many fish species in the Bull River and they are recognized as good indicators of river condition (Reice and Wohlenberg 1993, Boulton 1999, Norris and Thoms 1999, Norris and Hawkins 2000). Mayflies, stoneflies, caddisflies and chironomids were of particular interest because they are important food organisms for Bull trout, Westslope Cuttroat Trout (Schoby and Keeley 2011) and whitefish (McPhail and Troffe 1998) that are present in the Bull River (Cope 2005). Invertebrate counts by taxon and periphyton biomass were used in analyses to examine the effect of change in flow on assemblages in continually wetted areas of the diversion reach.

Cope (2005) showed that although peak discharge through the diversion reach could reach 120 $m^3 \cdot s^{-1}$, wetted width did not increase substantially as discharge rose above 40 $m^3 \cdot s^{-1}$ due to the steep banks constraining the box-shaped channel. Although the greatest absolute decrease in discharge through the diversion reach under the new spill schedule occurs at the peak of the freshet (from about 100 to 70 $m^3 \cdot s^{-1}$), the greatest exposure of river substrata occurs during the early ascending (March-April) and late descending limbs (July-September) of the annual hydrograph when the spill is far less than 40 $m^3 \cdot s^{-1}$. Of these two time periods, the optimum time for benthic production is July-September when disturbance by freshet has passed and temperatures are relatively high. Hence, benthic sampling occurred in August through September. Biological sampling in the diversion reach was restricted to the test segment where stream substrates are present that can host benthic assemblages.

To examine the effect of minimum flow on biological metrics, a Before-After-Control-Impact (BACI) paired layout was used (Stewart-Oaten et al. 1986). In this approach, the difference in value of a biological metric between a paired treatment and control site before change in flow was tested against the same difference after the change in flow. Sampling and measurements occurred in two years before the upgrade (2005 and 2006) and during three years after the upgrade (2009, 2010, and 2012). The sampling in 2009 occurred after commissioning of the upgraded hydropower facilities that occurred earlier that year. Years were replicates in this design, which means there were two replicates before the change in flow and three replicates afterwards. If the test of treatment effect was statistically significant, a conclusion was that change in flow modified benthic assemblages within the continually wetted habitat of the test segment. Conclusions were supported with multivariate analysis to examine change in whole assemblages over years and locations and to identify taxa that were most important in contributing to a flow effect, if it was present.

Three groups of measurements were completed at each site :

1. **Physical and chemical** variables that may be important in determining periphyton and invertebrate abundance and composition. Chemical data included all forms of nitrogen and phosphorus that are known to determine biological production in rivers in addition to basic analytes used to interpret water quality (temperature, dissolved oxygen, total dissolved solids, specific conductivity, pH, turbidity). Physical variables included flow, water velocity, water depth, wetted width, light attenuation, and approximate particle size distribution in the river substratum.

2. **Benthic invertebrate** measurements included abundance, richness, and community composition metrics. Samples were enumerated at the most reliable taxonomic level down to genus to support a range univariate and multivariate analyses.

3. **Periphyton** measurements included accrued biomass, richness, cell density, and biovolume by taxa. All periphyton measurements assisted with interpretation of space and time effects on the benthic invertebrate metrics.

2.2 Study Site

The Bull River drains an area of 1,530 km^2 on the west slope of the Rocky Mountains in British Columbia. The river originates in the Quinn Range at an elevation of 1,981 m and flows south, dropping 1,234 m over 21 km, to discharge into the Kootenay River at an elevation of 747 m near the town of Wardner. The climax forest of the study area consists of Engelmann spruce and subalpine fir but also includes Douglas-fir, western larch, and lodgepole pine. Riparian zones support these tree species and an understory of honeysuckle, saskatoon, spirea, false azalea, pinegrass, bunchberry, and mosses. The study area is within the Southern Continental Ranges Ecosection of the Southern Rocky Mountains Ecoregion of British Columbia (Demarchi et al. 1990). The Aberfeldie generating station is located 10.8 km upstream of the confluence of the Bull River with the Kootenay River, and the Aberfeldie Dam is situated 1.2 km upstream of the generating station. The headpond behind the dam has filled in with sediment over the 85 years of operation and water depths are estimated to be less than 2 m year round. Water residence time in the headpond is estimated to be 1-2 days at most. With this morphology, the headpond is similar to a large shallow pool in the river having a substrate of organic and inorganic sediment transported from upstream.

An upstream control site, called UBC1 (Upper Bull control site #1: 11U 622106 m E, 5495262 m N, 985 m ASL), was located 13.6 km upstream of the Aberfeldie Dam adjacent to the Bull River main line logging road (Figures 1 and 5). The sampled reach was characterized by flat run and riffle habitat. During non-rainy periods, the water is clear but turbidity can occur following heavy rainfall events in the headwaters and from occasional upstream slope failures. Westslope Cuttroat Trout (*Oncorhynchus clarkii*

lewisi), Rocky Mountain Whitefish (*Prosopium williamsoni*), Prickly Sculpin (*Cottus asper*) are the main fish species present in this reach of the Bull River (Cope 2005).

A treatment site, MBT1 (Middle Bull treatment site #1: 11U 618841 m E, 5483647 m N, 796 m ASL), was located 930 m downstream of the Aberfeldie Dam immediately downstream of the bedrock canyon within the test segment of the diversion reach (Figures 1, 3, and 4). The site was located upstream of the generating station and was accessed by a riverside trail that started from a parking area at the generating station. Sampling at MBT1 occurred in both of two channels. Substrate in the larger east channel had a large proportion of boulder while the west channel had a more equal mixture of large gravel, cobble, and boulder. Two B.C. blue-listed fish species, bull trout (*Salvelinus confluentus*) and Westslope Cutthroat Trout (*Oncorhynchus clarki clarki*), are known to be present at MBT1 along with brook trout (*Salvelinus fontinalis*), Kokanee (*Oncorhynchus nerka*), Rainbow Trout (*O. mykiss*), rocky mountain whitefish (*Prosopium williamsoni*), longnose dace (*Rhinichthys cataractae*), largescale sucker (*Catostomus macrocheilus*) and several sculpin species ((torrent sculpin (*Cottus rhotheus*), mottled sculpin (*C. bairdi*), and prickly sculpin (*C. Asper*)) (Cope 2005). Spawning Kokanee were present at MBT1 in all years of sampling (see cover photo). The canyon upstream of MBT1 functions primarily as overwintering and rearing habitat for fish entrained in spill over the dam. Leakage flows and the minimum winter flow of 0.25 $m^3 \cdot s^{-1}$ combined with deep pool refuge habitat support low densities of overwintering salmonids (Cope 2005).

A downstream site called LBC2 (11U 617865 m E, 5483386 m N, 786 m ASL), was located 4.2 km downstream of the dam and 1 km downstream of the generating station where flow was well mixed from the convergence of discharge from the powerhouse and flow from the diversion reach (Figures 1 and 5). The site was accessed via a 4-wheel drive road on the east side of the river or by wading from the west side when flows safely allowed. At this location, the river is a broad single channel where salmonids use rearing and spawning habitat.

UBC1

LBC2

Figure 5. Image of the Bull River at UBC1 (left) and LBC2 (right).

2.3 Flow

Daily mean flow in the Bull River at LBC2 was accessed from the Water Survey of Canada (WSC) for site number 08NG002 that is located between the generating station and LBC2 (Figure 1). Daily mean flow at UBC1 was determined as:

$$Q_i = Q_r(\frac{W_i}{W_r}) \qquad \text{Equation 1}$$

Where Q is daily mean flow ($m^3 \cdot s^{-1}$) at site i or the reference site r (WSC station 08NG002) and W is watershed area (km^2).

Mean daily flow releases to the diversion reach were provided by BC Hydro.

2.4 Field and laboratory procedures

2.4.1 Periphyton

Artificial substrata called "periphyton plates" were used to sample periphyton assemblages among years and sites (Figure 6). Each plate was a 30 x 30 x 0.64 cm sheet of open celled Styrofoam (Floracraft Corp. Pomona Corp. CA) attached to a plywood plate that was waterproofed with fibreglass resin and bolted to a concrete block. Styrofoam is a good substratum because its rough texture allows for rapid seeding by algal cells, and the adhered biomass is easily sampled (Perrin et al. 1987). Use of the artificial substrate standardized the substrate at all stations and removed variation in biomass accrual due to differences in roughness, shape, and aspect of substrates.

Figure 6. Image of a periphyton plate installed in the Bull River.

Periphyton biomass was sampled weekly from each of three periphyton plates that were installed at each site over an incubation period of seven weeks beginning in the first or second week of August each year. The plates were submerged in riffles. Depth and velocity over each plate was recorded on each visit. Each biomass sample consisted of a 2 cm diameter core of the Styrofoam and the adhered biomass that was removed as a punch from a random location on each plate using the open end of a 7 dram plastic vial. The sample was packed on ice and frozen at the end of each sampling day at -15ºC for later analysis. The last four weekly samples were analysed for concentration of chlorophyll-a (also called chl-a) using fluorometric procedures reported by Holm-Hansen et al (1965) and Nusch (1980). The highest chlorophyll-a concentration among these four samples from each plate was considered peak biomass (PB) for the sampling time series. PB always occurs in the final month of accrual of biomass on substrates installed in a river (Bothwell 1989). Samples collected over the weeks before the final four weeks were only analysed if anomalous PB values were found and additional data were needed to interpret accrual of biomass leading to PB. For example, if PB on a plate seemed exceptionally low, earlier samples from the time series could show the plate was disturbed, which would result in the PB value being discarded. Analysis of samples that are collected throughout a time series can also be used to derive an accrual curve. This analysis was not run in this project, but the samples were collected as general practice in case such an analysis was needed or later requested as an aid to interpretation of periphyton production.

On the final periphyton sampling day, one additional core was removed from each plate and preserved in Lugol's solution. Biomass was removed from the Styrofoam punch using a fine spray from a dental cleaning instrument within the sample vial.

Contents were washed into a graduated and cone shaped centrifuge tube and water was added to make up a known volume. The tube was capped and shaken to thoroughly mix the algal cells. An aliquot of known volume was transferred to a Utermohl chamber using a pipette and allowed to settle for a minimum of 24 hours. Cells were counted along transects examined first at 300X magnification to count large cells and then at 600X magnification to count small cells under an Olympus CK-40 inverted microscope equipped with phase contrast objectives. Only intact cells containing cytoplasm were counted. A minimum of 100 cells of the most abundant species and a minimum of 300 cells were counted per sample. The biovolume of each taxon was determined as the cell count multiplied by the volume of a geometric shape corresponding most closely with the size and shape of the algal taxon. Data were expressed as number of cells and biovolume per unit area of the Styrofoam punch corrected for the proportion of total sample volume that was examined in the Utermohl chamber.

2.4.2 Benthic invertebrates

Benthic invertebrates were sampled using a Surber net (Merritt et al. 1996) at the time of final periphyton sampling in late September of each year using methods consistent with provincial standards (Cavanagh et al. 1997). The sampler had a surface area of 900 cm^2 and was equipped with a 250 µm mesh Nitex collection net and removable cod end (Figure 7). At each site the sampler was placed at a randomly selected location. Substrate within the sampling frame was disturbed to a depth of 10 cm for a period of one minute using a garden fork. The sampler was then moved upstream roughly 1.5 m to another undisturbed location and the sample collection was repeated. Contents accumulated in the net after five placements constituted a single sample. Total surface area for a single sample was 4,500 cm^2 (900 cm^2 sampler area x 5 locations). Four samples were collected at each of the three river sites each year. The samples were preserved in 10% formalin immediately after collection.

Figure 7. Image of Surber net being used for collection of benthic invertebrates in the Bull River.

In the laboratory, each benthos sample was washed through 1 mm and 250 µm mesh sieves to yield a macrobenthos fraction (>1 mm) and a microbenthos fraction (<1 mm and >250 µm). Animals were picked from twigs, grasses, clumps of algae, and other debris and returned to the 1 mm sieve. Microbenthos was split into 16 subsamples using a plankton splitter. Animals were enumerated from successive sub-samples until 200 animals were counted. If 200 or fewer animals were counted part way through the sorting of a sub-sample, that entire sub-sample was sorted. The macrobenthos fraction was enumerated in its entirety. Sub-sample counts were extrapolated to the total sample. The total sample count was the sum of microbenthos and macrobenthos in the complete sample. The animals were identified to the most reliable taxonomic level down to genus using keys from Edmondson (1959), Merritt and Cummins (1996), and Pennak (1978). One in 10 samples was sorted twice to test efficiency of the first sort. A target for acceptable sorting was that 90% of the sample must be enumerated on the first sort. If efficiency was <90%, samples in the group to which the test applied were re-sorted. Sorting efficiency was >90% on the first sort of all samples.

2.4.3 Habitat Measurements

Habitat measurements were completed for purposes of describing habitat conditions in the riffles from which the periphyton and benthic invertebrates were collected. Water temperature was logged in two hour intervals using an Onset Hobo

logger (Onset Computer Corp, Pocasset MA) that was attached to a periphyton plate at each site. The logger had an accuracy of ±0.2ºC over a temperature range of 0-50ºC (Onset Computer Corporation, Part number MAN-U22-001, Doc#: 10366-A Specifications). A Wolman pebble count (Wolman, 1954) was done at each site each year, in which the intermediate diameters of 100 randomly-selected stones at each site were measured using a gravelometer (Wildco, Buffalo NY). There is no variation in accuracy or precision with this equipment. It is a direct measure of size of opening through which a stone will pass. Median particle size (D_{50}) was calculated from the pebble data as the median size among all 100 stones. Water samples were collected from each site at the start and finish of the periphyton sampling for analysis of soluble reactive phosphorus, total phosphorus, total dissolved phosphorus, particulate phosphorus, ammonium-N, and nitrate-N concentration. All samples for analysis of dissolved fractions were filtered in the field through pre-ashed 0.45 μm GF filters using a Swinnex syringe filtration apparatus (EMD Millipore, Billerica, MA, USA). The samples were delivered to the Fisheries and Oceans lab at Cultus Lake for analysis. This lab specializes in analysis of nutrients at low detection limits. Samples for the total phosphorus and total dissolved phosphorus analyses were digested and analysed using Menzel and Corwin's (1965) potassium persulfate method. soluble reactive phosphorus was analysed using the molybdenum blue method (Murphy and Riley 1962). Particulate P was determined by difference between total phosphorus and total dissolved phosphorus. ammonium-N and nitrate-N were analysed using a Technicon autoanalyzer (Stainton et al. 1977). The sum of ammonium-N and nitrate-N was called dissolved inorganic nitrogen.

The Cultus lab reports precision, expressed as percent of two times standard deviations around a standard as 1.9 – 4.4% for total phosphorus, 2.7 – 6.7% for soluble reactive phosphorus, 1.0 – 3.2% for ammonium-N, and 0.7 – 11.4% for nitrate-N. The lab also reports ranges of percent recovery of known standards as 97 – 101% for total phosphorus, 96 – 99% for soluble reactive phosphorus, 97 – 102% for ammonium-N, and 97 – 103% for nitrate-N. These data are from personal communications with the senior lab technician at Cultus (K. Parish, Fisheries and Oceans Canada, Cultus Lake Lab, personal communication, Oct 16, 2012).

One blank water sample was processed on each day of sampling to provide information on contamination from handling and one blind duplicate sample (no site label) was collected each day to estimate field sampling precision. Each blank and duplicate was analysed for each chemical parameter. Blanks were double deionized water samples provided by the Cultus Lake lab and handled the same way as all test samples including filtration, water transfers to sample bottles, storage in the fridge or freezer, and shipping. The presence of analytes in the blank samples indicated contamination during sample processing and the chemical concentration showed the amount of contamination. Field precision (D_f) was calculated as relative percent difference of an analyte concentration between a sample and its corresponding duplicate

using the following equation recommended by the Ministry of Environment Lands and Parks (1988):

$$D_f = \left(\frac{A-B}{(A+B)/2}\right) * 100 \qquad \text{Equation 2}$$

where A is the concentration of an analyte in sample A and B is the concentration of the same analyte in the duplicate sample.

A YSI 6920 Sonde calibrated with fresh standards on the evening before use was used for measurements of turbidity, total dissolved solids concentration, dissolved oxygen concentration, and pH at the start and finish of the sampling time series each year.

Digital photographs were taken at the end of the sampling series to provide a visual record of site conditions.

2.5 Data analysis

2.5.1 Quality assurance

Following original data entry by the second author the senior author performed quality checks. Every tenth row of each data sheet that appears as a digital appendix to this report was checked against raw field data sheets or lab reports. If errors were found, the second author checked complete sections of data where an error occurred. Uncertainties found by the senior author related to units or other labelling were either corrected by the senior author or the sent back to the junior author to make corrections and improve clarity. All data anomalies (values very different from others) were highlighted and checked for accuracy. If no explanation could be found for obvious outliers, those outlying data were removed from the data set.

Statistical analyses were initially run by the junior author. Output was checked by the senior author and requests for clarification were sent back to the junior author. During the data analyses, different approaches were discussed extensively both between the authors and with external specialists. Final analyses were the result of agreement between all people as to what would be the best approach to answer the management questions.

2.5.2 Water release to the diversion reach

Flow curves from all sites were examined for trends and anomalies. Flow over time in the diversion reach was examined to determine if the flow release from the dam met the prescribed criteria. Mean and median daily flows for the periods of biological sampling each year (August through early October) were calculated to describe the

magnitude of change in flow in the diversion reach between the “before” and “after” years.

2.5.3 Description of physical and chemical variables in the Bull River

For descriptive purposes, mean values of the physical and chemical variables were calculated for each site in the time periods before and after the implementation of minimum flow.

2.5.4 Univariate tests of the effect of minimum flow in the diversion reach on benthos metrics

The effect of change in flow in the diversion reach at MBT1 on benthic invertebrates was tested on each of five metrics: counts of all benthic invertebrates (all taxa), the sum of counts of Ephemeroptera, Plecoptera and Trichoptera (EPT) , counts of chironomids, invertebrate richness at the most reliable taxonomic level down to genus, and Simpson’s Index of diversity of invertebrates. Genus richness is richness of taxa identified to the lowest reliable level where genus was the lowest level of identification. Simpson’s Index is a measure of community heterogeneity and tends to weight common taxa more than rare taxa (Krebs 1999). This measurement provided a contrast to taxonomic richness that weights all taxa evenly. The effect of change in operations at LBC2 was tested on the same five metrics. This latter analysis was not a test of change in flow because flow did not change at LBC2 over time. It was a test of operations because different entrainment velocities associated with the new intake to the penstock could modify nutrient and sediment transport from the headpond and affect habitat supporting biological communities in the lower Bull River at LBC2 differently than was occurring before the upgrade.

In both cases (test of a flow effect at MBT1 and a test of an operations effect at LBC2), a one way analysis of variance (ANOVA), also known as a Students t-test, was run using the BACI layout reported by Stewart-Oaten et al. (1986) to determine if there was a significant effect of flow on each of the five metrics. Detection of an effect of change in flow was achieved by testing whether the difference between metric values at UBC1 (control site) and MBT1 (impact site) changed between the before and after time periods. Similarly, detection of an effect of change in operations in the downstream reach was achieved by testing whether the difference between metric values at UBC1 (control site) and LBC2 (impact site) changed after the upgrade. The average value of a given metric was determined from the four benthic invertebrate samples that were collected at each site and time combination to avoid pseudoreplication (multiple samples within a site are not true replicates; Hurlbert 1984). The difference of that value between a potentially disturbed site (e.g. MBT1 or LBC2) and the control site (UBC1) was determined for each year. The t-test showed whether there was a statistically significant

difference between the mean paired difference from the "before" years and the mean paired difference from the "after" years. The tests were run in Systat v11 (Systat 2004) with α set at 0.1. This level of α is higher than the typical default of 0.05 because a 90% probability that two sample distributions are different was acceptable for purposes of determining if an effect of flow or an effect of change in operations on benthos metrics was present. The count data were $\log_{10}(X+1)$ transformed prior to analysis to better meet the assumptions of equal variance and normal distribution needed for ANOVA. A significant difference ($p<0.1$) meant that the difference of the mean metric value between site pairs after the change in flow was different from the mean metric value between site pairs before the change in flow. This difference was attributed to an effect of change in flow at MBT1 or a change in operations at LBC2. There was no change in flow at LBC2 but increased entrainment of sediment and associated nutrients from the headpond into the new penstock due to increased rates of water withdrawal compared to rates before the upgrade may influence biological production downstream of the powerhouse where all water is returned to the main river channel.

If the test showed no significant difference ($p>0.1$) and thereby no effect of flow or no effect of change in operations on the metric being tested, a post-hoc power analysis was run in Systat 11 to determine the probability of obtaining a non-significant result if an effect was actually present (a Type II error), as recommended in recent water monitoring guidelines by Environment Canada (Environment Canada 2012).

2.5.5 Multivariate tests of temporal and spatial change in benthic invertebrate and periphyton assemblages

Multivariate tests to examine space and time effects on biological assemblages were run because they can be more comprehensive than univariate ANOVA's by including all taxa that are identified in an assemblage rather than a single metric. Invertebrate count data from "before" (2005 and 2006) and "after" (2009, 2010 and 2012) years were compiled with time (before and after change in flow), year, and location (UBC1, MBT1 and LBC2) coded as factors to facilitate several types of analysis. Terrestrial dwelling taxa and adult stages of aquatic insects were removed from the dataset because they did not represent individuals known to be rearing in water at the sampling site. The abundance of some taxa was high in some samples but low in others at the same location and time. This clumping could mask temporal or spatial signals. To reduce the influence of these taxa on assemblage patterns, the variance to mean ratio among replicate samples at each site and time (*i.e.,* for a combined factor, siteyear) was averaged to derive an index of dispersion (*D*) for each taxon (Clarke and Gorley 2006). *D* values near 1 indicated no clumping while larger values, particularly >10 and certainly >100 showed the presence of clumping. A frequency plot was used to examine the distribution of *D* among taxa. Weighting was carried out by dividing the counts for each taxon by *D*. No further transformation was applied following dispersion weighting. The

difference between dispersion weighting and other common transformations used in multivariate statistics to weight rare or common taxa (e.g., square root or fourth root or log) is that dispersion weighting targets individual taxa that have particularly high variance while the other transformations are broad spectrum procedures that affect all taxa the same way.

To avoid pseudoreplication, the average value of a given taxon count was determined from the four benthic invertebrate samples that were collected at each site and time combination. By doing so, the number of observations changed from 60 (5 years x 3 sites x 4 samples) to 15 (5 years x 3 sites x 1 average value).

A non-metric multi-dimensional scaling analysis (NMDS) was run on a ranked Bray-Curtis similarity matrix of dispersion-weighted count data to examine dissimilarities of assemblages between all combinations of location and year (5 years x 3 sites = 15 observations). NMDS is an ordination technique for fitting a set of points in space such that the distances between points correspond as closely as possible to dissimilarities between them. A 'stress value' measures distortion of the multidimensional data on the 2D plot. The ordination was considered usable if it had a stress value <0.2, following recommendations by Clarke and Gorley (2006). The ordination was used to examine temporal and spatial patterns in the count data.

Another multivariate statistical tool called RELATE (Clarke and Gorley 2006) was used to determine if a significant upstream to downstream gradient among benthos and periphyton assemblages was present in the Bull River from UBC1 to LBC2. The test was run on a ranked Bray-Curtis similarity matrix of dispersion weighted count (invertebrates) or biovolume (periphyton) data. The statistic *p* (rho) showed the extent to which assemblages among samples followed a linear series (e.g., a test of seriation) in each year. Rho values close to zero show little or no gradient while values close to 1 show that assemblages change over a linear gradient.

Location and time effects were tested on the multivariate data by two-way crossed analysis of similarity (ANOSIM) on the same dispersion weighted count data used in NMDS. Similar to the BACI t-test layout, two separate analyses were run, the first contrasting UBC1 and MBT1 and the second contrasting UBC1 and LBC2. Each of the site and year effects were interpreted from the ANOSIM R statistic that varies from 0 (no site or year effect on assemblages) to 1 (dissimilarities of assemblages between sites or years were greater than dissimilarities within site or year). Significance was tested by standard permutation in Primer (Clarke and Gorley 2006). The multivariate similarity percentages procedure called SIMPER, also run in Primer v6 (Clarke and Gorley 2006), was used to identify invertebrate genera or higher classification cumulatively contributing to >90% of similarities of assemblages between sites and times. One SIMPER analysis was run for the contrast between UBC1 and MBT1 between the two times and a second analysis contrasted UBC1 and LBC2.

3 RESULTS

3.1 Water Release to the Diversion Reach

Flows in the diversion reach were different among years before and after the upgrade (Table 1, Figure 8). Among the "before" years, mean daily flow in the diversion reach was 2.4 $m^3 \cdot s^{-1}$ to 24 $m^3 \cdot s^{-1}$ with peaks in late August and late September. Minimum flow was implemented in 2009 but within five days of starting the biological sampling, all Bull River flow was released from Aberfeldie Dam into the diversion reach to accommodate 20 days of maintenance in the penstock and powerhouse. The study team was given no notice of this offline period and no changes could be made to schedules to avoid sampling during the water release. The result was a 20-day period at the beginning of sampling in August 2009 when flows reached 45 $m^3 \cdot s^{-1}$ in the diversion reach followed by the minimum flow release of 2 $m^3 \cdot s^{-1}$ during the following 29 days of sampling. The anomalous water release in 2009 caused mean flow in the diversion reach during the 2009 sampling period (16.5 $m^3 \cdot s^{-1}$) to be greater than mean flow during sampling periods before the upgrade (10.3 $m^3 \cdot s^{-1}$ in 2005, 4.5 $m^3 \cdot s^{-1}$ in 2006, Table 1). In 2010, the minimum flow was maintained throughout the sampling period except for two days at the start of the sampling period (4.2 $m^3 \cdot s^{-1}$ on the first day and 7.2 $m^3 \cdot s^{-1}$ on the second day), and during six days near the end of sampling when flows reached 36 $m^3 \cdot s^{-1}$. In 2012, there were four days in mid-September when flows were greater than 3 $m^3 \cdot s^{-1}$, otherwise the 2 $m^3 \cdot s^{-1}$ minimum flow was maintained through the sampling period. The coefficient of variation of flow during the sampling periods was <0.5 in the pre-upgrade years and in 2012, but was greater than 1 in 2009 and 2010 (Table 1).

Exclusion of 2009 from calculations of mean flow after the upgrade resulted in a mean flow of 3.3 $m^3 \cdot s^{-1}$, which was closer to the target flow to be tested of 2 $m^3 \cdot s^{-1}$ than if 2009 flow data were included. This post-upgrade flow was 56% lower than mean daily flow in the pre-upgrade period (7.5 $m^3 \cdot s^{-1}$). Using the same data, median flows pre- and post-upgrade were 6.3 $m^3 \cdot s^{-1}$ and 2.0 $m^3 \cdot s^{-1}$ respectively. Frequency distributions of flows before and after the upgrade, not including 2009, show a marked shift to lower flows after the upgrade, with 90% of all mean daily flows being at or within 1 $m^3 \cdot s^{-1}$ of the prescribed minimum flow (Figure 8).

Table 1. Statistics summarizing water release to the diversion reach during the sampling period (August and September) before and after the implementation of minimum flow.

Year	Time period (before or after the upgrade)	Mean daily flow during August to September sampling in the diversion reach ± SD ($m^3 \cdot s^{-1}$)	Median flow in the diversion reach ($m^3 \cdot s^{-1}$)	Coefficient of variation	Range of mean daily flow ($m^3 \cdot s^{-1}$)
2005	Before	10.3 ± 4.6	9.6	0.45	3.7 – 23.5
2006	Before	4.5 ± 2.0	3.7	0.45	2.4 – 10.1
2009	After	16.5 ± 17.4	2.0	1.06	2.0 – 44.8
2010	After	4.3 ± 7.7	2.0	1.80	0.5 – 36.0
2012	After	2.1 ± 0.9	2.0	0.42	0.5 – 6.2
2005 and 2006	Before	7.5 ± 4.6	6.3	0.62	2.4 – 23.5
2009, 2010, and 2012	After (all years)	7.5 ± 12.5	2.0	1.67	0.5 – 44.8
2010 and 2012	After (excluding 2009)	3.3 ± 5.7	2.0	1.75	0.5 – 36.0

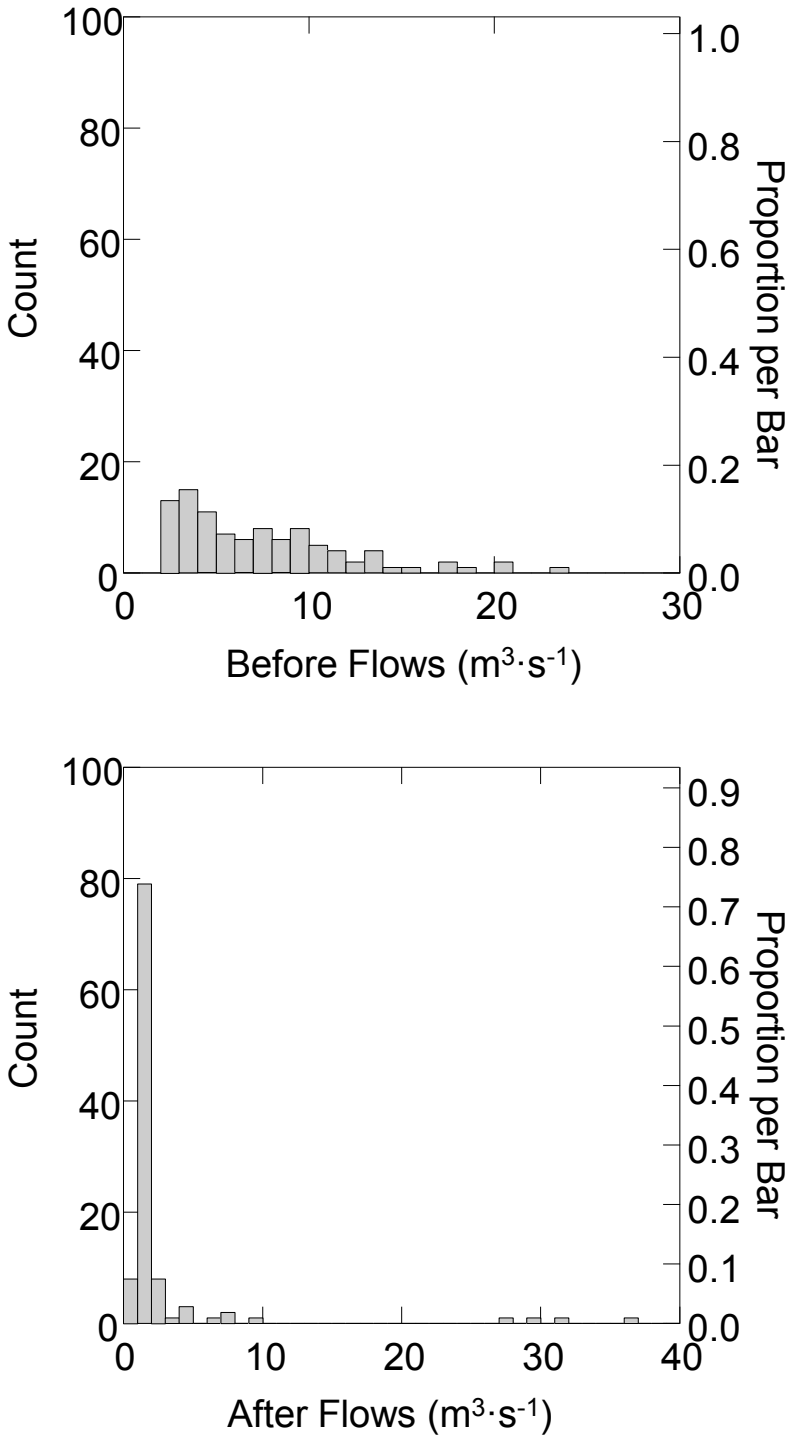


Figure 8. Frequency distribution of mean daily flow during August and September sampling periods in the diversion reach before (top) and after (bottom) the Aberfeldie upgrade. The after data does not include 2009 when an anomalous flow release to the diversion reach occurred.

3.2 Physical and Chemical Variables

3.2.1 Quality of chemical analysis of water samples

Relative percent differences between replicate pairs of samples ranged between 2% and 38% (Table 2). Precision is considered high when relative percent difference is less than 25% (Ministry of Environment Lands and Parks (1988)). This high precision was found for total phosphorus, total dissolved phosphorus, and nitrate-N but not for soluble reactive phosphorus (28%) and ammonium-N (38%). Some variability between replicate water samples is expected not only related to sample handling and processing but due to natural variability captured in the separate water samples. Low precision is expected for soluble reactive phosphorus and ammonium-N because they occurred in extremely low concentrations, approaching the method detection limit for each test (Table 3) at which point error can increase. For ammonium-N, additional confounding at extremely low concentrations can be caused by absorption of ammonia from the air during sample filtrations. Scavenging of ammonia from the air resulted in 70% of the blanks showing positive ammonium. This scavenging is caused by low pH of double deionized water. It does not occur as much in natural river samples having circumneutral pH, which means that analysis of blanks is not an effective QA test for analysis of ammonium. The occurrence of nitrate-N in blanks was only 4% of average nitrate-N concentrations, which was considered too low to be a factor influencing interpretations of the nitrate-N data. For soluble reactive phosphorus, we found three times the method detection limit in 20% of the sample replicates, representing 40% of soluble reactive phosphorus concentration in the river water samples. Given that high variability, we conclude there is little confidence in soluble reactive phosphorus results less than 1 $\mu g \cdot L^{-1}$, which happens to the method detection limit for soluble reactive phosphorus at commercial labs.

Table 2. Relative percent differences of analyte concentrations between sample replicates from the field.

Analyte	**Average value (± sd) of relative percent difference between samples and their duplicates (%)**
Total phosphorus	13.2 ± 6.3 (n=9)
Total dissolved phosphorus	12.7 ± 9.1 (n=10)
Soluble reactive phosphorus	27.9 ± 25.5 (n=10)
Ammonium-N	37.5 ± 16.6 (n=10)
Nitrate-N	2.4 ± 2.4 (n=10)

Table 3. Incidence of positive blanks (blanks having an analyte concentration above the method detection limit) and comparison of analyte concentrations in positive blanks with those in river samples.

Analyte	Method detection limit ($\mu g \cdot L^{-1}$)*	Number of positive blanks (maximum possible is 10 (two samples per year over five years))	Average concentration in positive blanks ($\mu g \cdot L^{-1}$)	Average concentration in river samples ($\mu g \cdot L^{-1}$)
Total phosphorus	0.2	1	1.5	9.6
Total dissolved phosphorus	0.2	2	1.3	4.1
Soluble reactive phosphorus	0.1	2	0.3	1.2
Ammonium-N	1.0	7	2.8	3.7
Nitrate-N	0.1	5	2.6	60

*Method detection limit is defined by the Cultus Lake lab as the smallest detectable signal.

3.2.2 Field and lab data

Physical and chemical data showed the Bull River to be pristine (Table 4). The river had high concentrations of dissolved oxygen. Mean daily temperatures were 9.5ºC to 10.5ºC, and they increased by less than 0.6°C between UBC1 and LBC2 (Table 4). Mean daily temperature was 0.3ºC to 0.4ºC cooler at all sites in the years following implementation of minimum flow. Total dissolved solids concentration was 190 – 216 $mg \cdot L^{-1}$, which is expected in drainages from sedimentary formations of the Rocky Mountains. Concentrations of dissolved inorganic nitrogen were 58-65 $\mu g \cdot L^{-1}$ and did not vary by site or time. Concentrations of soluble reactive phosphorus were 0.5 – 0.6 $\mu g \cdot L^{-1}$ before the upgrade and 1.6 – 1.9 $\mu g \cdot L^{-1}$ after the upgrade with little variation between sites in either time period. Average turbidity was <2.7 NTU among all sites and times. Mean turbidity was higher after the upgrade than before but standard deviations far exceed the means of both time periods, which showed the means would not be statistically different.

The molar ratio of bioavailable nitrogen:phosphorus in water can indicate whether nitrogen (N) or phosphorus (P) potentially limits algal growth. Either of these two nutrients will almost universally limit algal growth (Wetzel 2001). The ratio is calculated from bioavailable forms of N, which includes dissolved inorganic nitrogen (ammonium-N plus nitrate-N) and P, which is best represented by soluble reactive phosphorus when it can be detected as was the case in the Bull River. Rhee (1978) showed that for a given species of algae, there is a sharp transition between P-limited and N-limited growth. The particular N:P ratio (using bioavailable forms of N and P) at which the transition between N and P-limitation occurs is species dependent, varying from as low as 7:1 for some diatoms (Rhee and Gotham 1980) to as high as 45:1 for some blue-green algae (Healey 1985). Guildford and Hecky (2000) found that N-deficient growth of algae occurs at molar nitrogen:phosphorus <20 while P-deficient growth occurs at nitrogen:phosphorus >50. At intermediate ratios, either N or P can be deficient. The

molar nitrogen:phosphorus ranged from 65 to 402:1 in the Bull River with higher ratios in the "before" years than the "after" years, all being in the range that indicates phosphorus limitation of algal growth.

Median pebble size (D_{50}) was greater at MBT1 than at UBC1 or LBC2.

Table 4 Mean temperature and concentration or measure of chemical parameter at the three Bull River sites in the years before (2005 and 2006) and after (2010 and 2012) implementation of minimum flow in the diversion reach (MBT1).

Parameter	Time	N	Measure at UBC1 ± SD	Measure at MBT1 ± SD	Measure at LBC2 ± SD
Water temperature (C)	Before		9.9 ± 2.1	10.5 ± 2.1	10.5 ± 2.0
	After		9.5 ± 1.6	10.2 ± 1.7	10.1 ± 2.3
Soluble reactive phosphorus ($\mu g \cdot L^{-1}$)	Before	4	0.53 ± 0.36	0.64 ± 0.25	0.64 ± 0.30
	After	4	1.77 ± 0.75	1.89 ± 0.55	1.62 ± 0.81
Dissolved inorganic nitrogen ($\mu g \cdot L^{-1}$)	Before	4	65.2 ± 16.2	62.0 ± 9.0	62.3 ± 12.7
	After	4	63.0 ± 21.1	65.1 ± 24.3	58.1 ± 20.6
Molar N:P ratio	Before	4	402	232	278
	After	4	110	65	118
Median Pebble Size D_{50} (mm)	Before	2	180	250	180
	After	2	180	> 300	180
Dissolved Oxygen ($mg \cdot L^{-1}$)	Before	4	10.0 ± 0.7	10.6 ± 0.7	10.0 ± 0.7
	After	15	10.6 ± 0.5	10.8 ± 0.6	10.6 ± 0.5
Total dissolved solids ($mg \cdot L^{-1}$)	Before	4	216 ± 19	210 ± 19	205 ± 10
	After	15	197 ± 19	185 ± 26	190 ± 21
Turbidity (NTU)	Before	4	1.1 ± 1.3	1.3 ± 0.9	1.2 ± 1.2
	After	15	2.0 ± 4.5	2.1 ± 4.2	2.6 ± 6.9
pH	Before	4	7.5 ± 0.2	8.2 ± 0.3	8.1 ± 0.2

Parameter	Time	N	Measure at UBC1 ± SD	Measure at MBT1 ± SD	Measure at LBC2 ± SD
	After	15	8.1 ± 0.5	8.4 ± 0.2	8.3 ± 0.1

3.3 Benthic Invertebrates

3.3.1 Density and composition

The composition of benthic invertebrates was similar between sites and times (Figure 9). Most genera or higher taxa were from the Diptera (15 taxa) followed by Ephemeroptera (mayflies, 13 taxa), Plecoptera (stoneflies, 13 taxa), Trichoptera (caddisflies, 10 taxa) and Acari (mites, 9 taxa). Mayflies accounted for 24% to 52% of the assemblages, with most being *Baetis* sp, *Ephemerella* sp,, *Rhithrogena* sp., *Diphetor,* and *Cinygmula* sp. Chironomids accounted for 16 to 50% of individuals and other dipterans, most commonly *Antocha* and *Wiedemannia*, added another 1% to 7% to total numbers. Up to 24% of the animals were stoneflies, with most common taxa being young Capniidae, *Zapada* sp., Nemouridae, *Sweltsa* and *Taenionema* sp. Although a diverse group of trichopterans was found, densities were low (1 to 4% of the total number of animals) with most common taxa being *Arctopsyche*, *Rhyacophila* and *Hydropsyche*. Important food organisms for fish in the Bull River (mayflies, stoneflies, caddisflies, and chironomids) accounted for 77 to 90% of animals at all sites. The remainder was comprised of mites (Torrenticolidae, *Lebertia* sp. and *Sperchon*) and naidid worms.

Mean density of all animals was lowest at UBC1 and highest at LBC2 before the upgrade but afterwards, the highest density was found at MBT1. Downstream of the powerhouse, there was a larger proportion of chironomids and a smaller proportion of mayflies than were found upstream. Among the “after” years, mean invertebrate density decreased by 23% at UBC1 and by 8% at LBC2, while it increased by 61% at MBT1 compared to densities among the “before” years (Figure 10). The increase at MBT1 was mainly due to mayflies and chironomids.

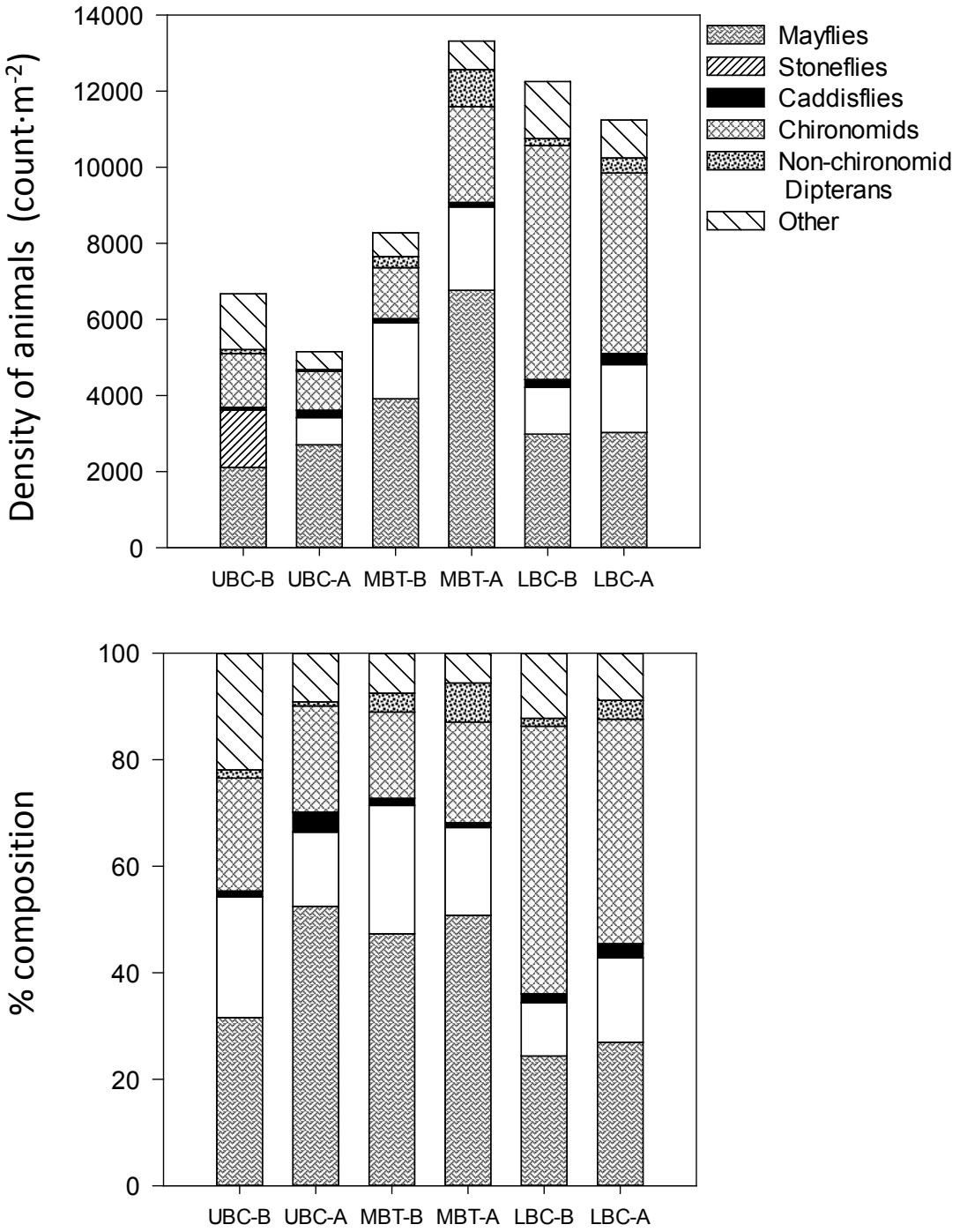


Figure 9 Mean density (top) and composition (top and bottom) of invertebrate communities at UBC1, MBT1, and LBC2 before and after the upgrade. The "A" following the site name means after the upgrade and 'B" means before the upgrade. "Other" includes Coleoptera, Collembola, Haplotaxida, Hemiptera, Acari, Lumbriculida, Nemata, and Ostracoda.

3.3.2 Tests of the effect of minimum flow in the diversion reach

The t-test run in the BACI layout showed no significant effect (p>0.1) of the change in flow on total density, density of chironomids, richness at the most reliable taxonomic level down to genus, and Simpsons Diversity in the diversion reach at MBT1 (Table 5). Similarly, there was no significant effect (p>0.1) of the change in operation on all metrics in the downstream reach at LBC2. The former test did not include 2009 in the "after" years because of anomalous flows above the level to be tested in that year (Section 3.1).

In contrast, the change in flow increased the abundance of EPT in the diversion reach at MBT1 (p<0.1). EPT density was unchanged between the "before" and "after" periods at UBC1 but it increased by 50% at MBT1. Figure 12 shows that this divergence was mostly due to high densities in 2010 compared to the other years. Figures 10, 11, 13 and 14 show the considerable overlap of variance around mean values of the other metrics between sites by year and the lack of a change in values between UBC1 and MBT1 between the "before" and "after" years.

Power was calculated for tests that showed no significant effect of flow on a given metric. Power is irrelevant for tests that do show an effect of flow because an effect has been detected so there is no interest in the probability of detecting one. Power of the tests of effects of change in flow on benthos in the diversion reach at MBT1 ranged from 0.05 to 0.1 (Table 5) and power of the test of effects of change in operation on benthos metrics in the downstream reach at LBC2 ranged from 0.05 to 0.11 (Table 6). These results mean there was a little chance of detecting an effect of flow on benthos at MBT1 (not including the EPT) and at LBC2 if there was one.

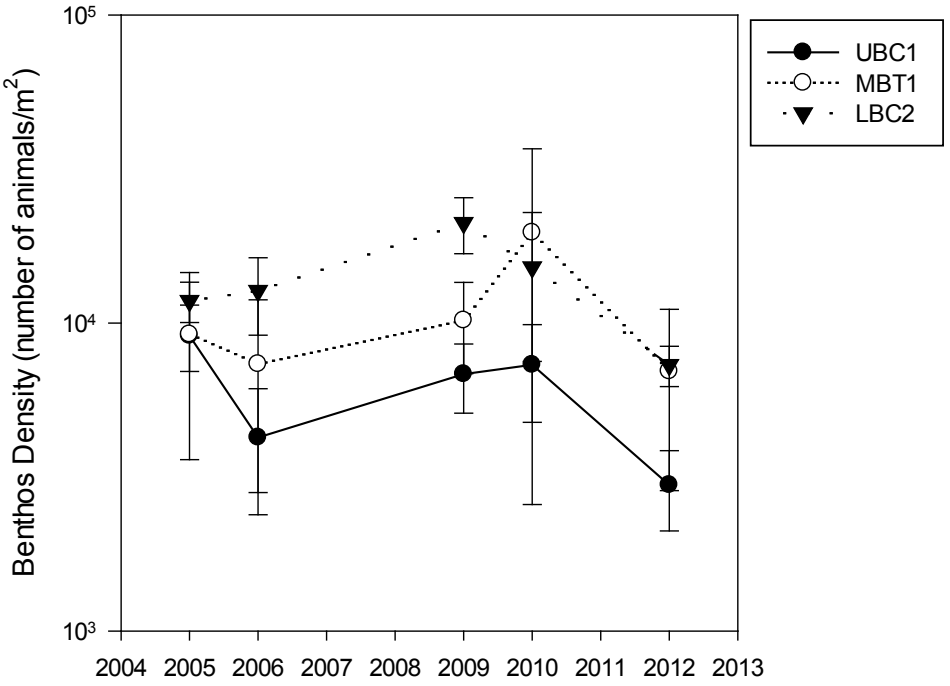


Figure 10 Mean density (±standard deviation) of all invertebrates among sites before and after the upgrade.

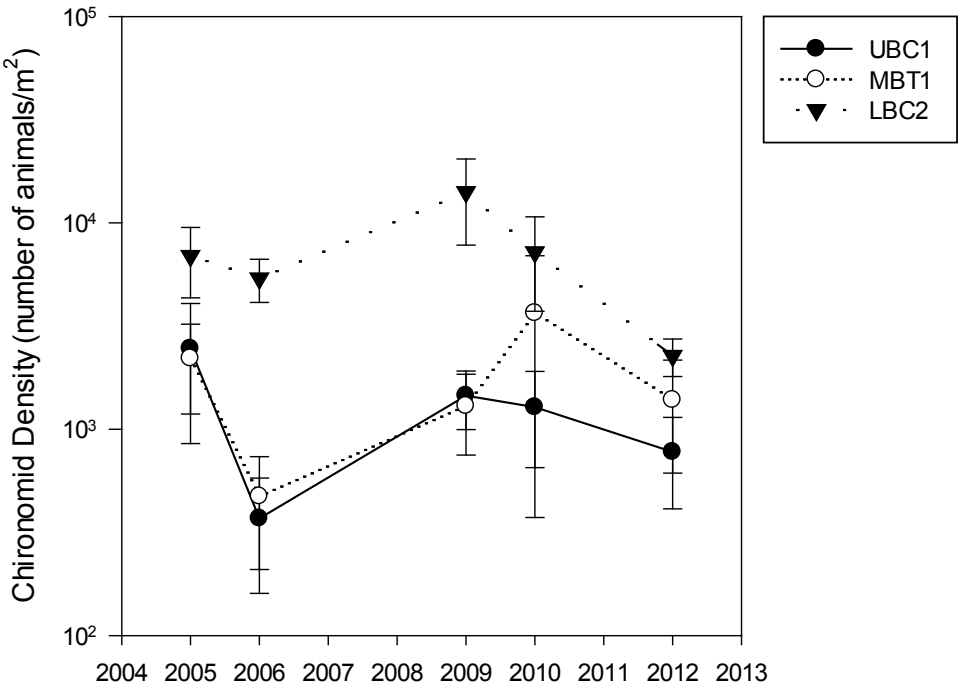


Figure 11 Mean density (±standard deviation) of chironomids among sites before and after the upgrade.

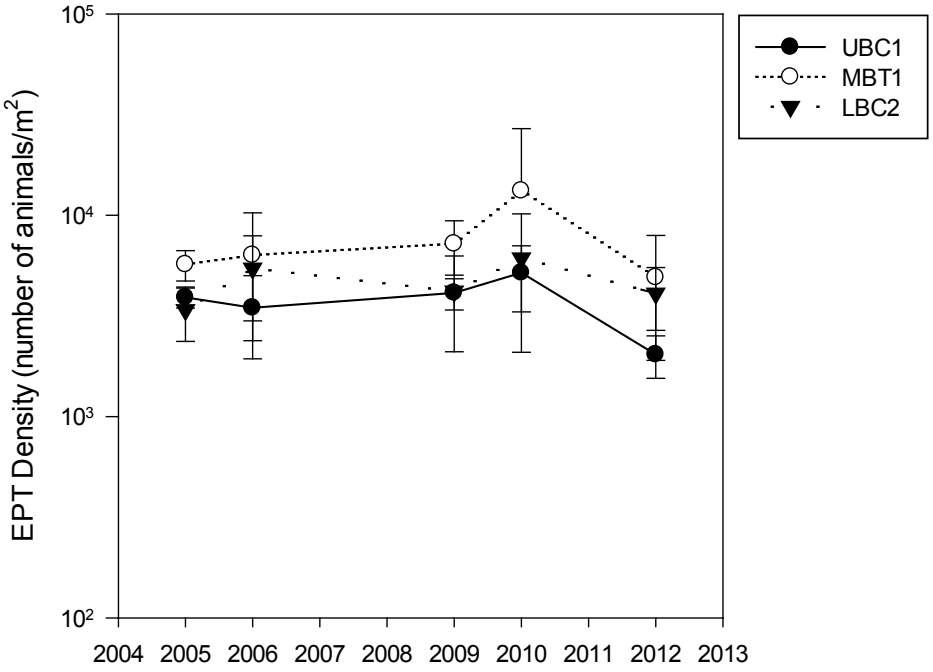


Figure 12 Mean density (±standard deviation) of the EPT among sites before and after the upgrade.

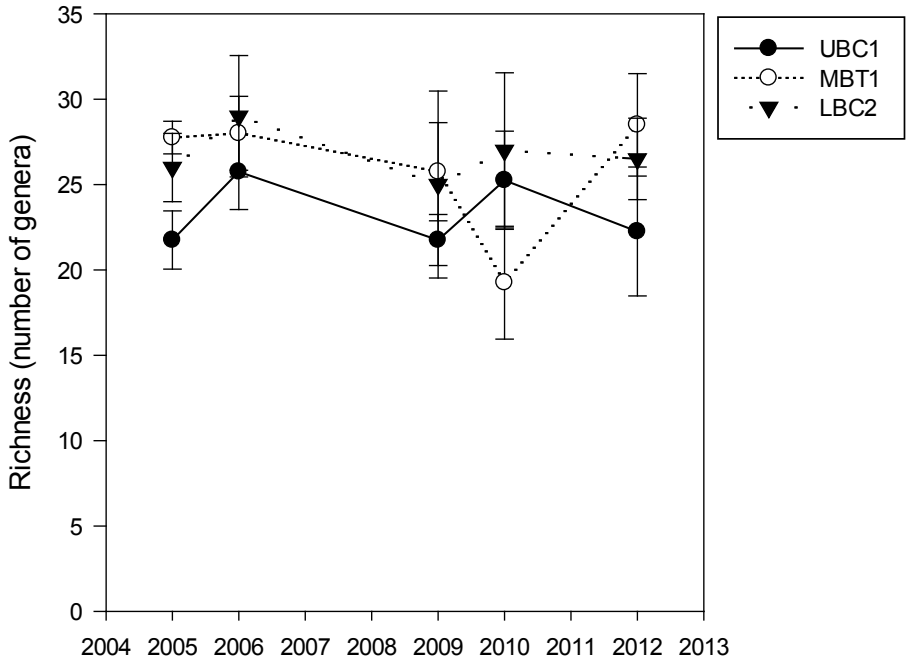


Figure 13 Mean richness at the most reliable taxonomic level down to genus (±standard deviation) among sites before and after the upgrade.

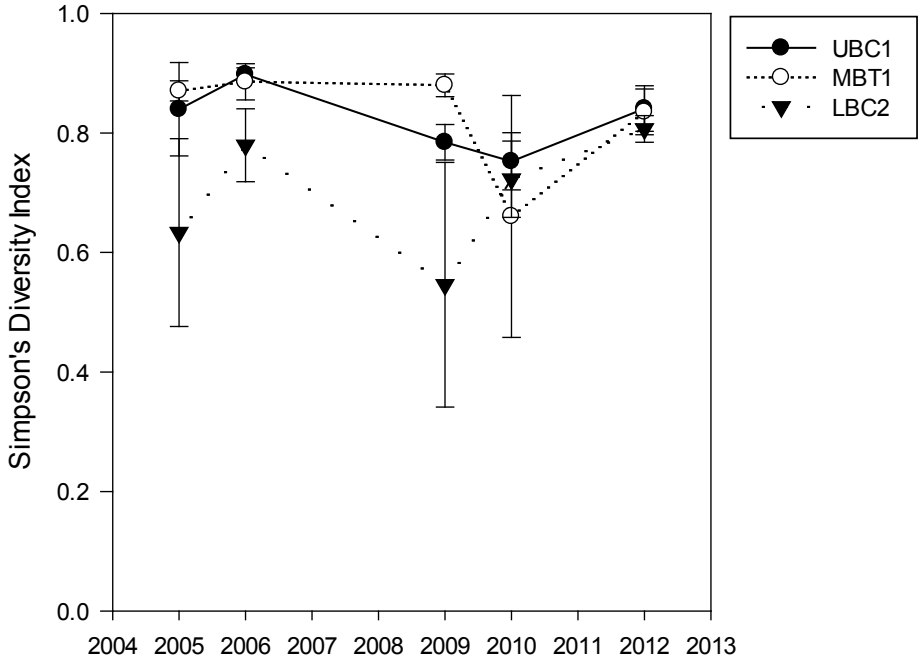


Figure 14 Mean value of Simpson's Index of Diversity (±standard deviation) among sites before and after the upgrade.

Table 5. Mean difference in value of benthic invertebrate metrics (±SD) between UBC1 and MBT1 before and after the upgrade. Data from 2009 were not included in the "after" data because flow in that year exceeded the minimum flow to be tested.

Metric	Time	N	UBC1	MBT1	Difference between means at MBT1 and UBC1	*p*	Post-hoc power of the test
Total abundance of animals (number·m^{-2})	Before	2	6670 ± 3421	8273 ± 1301	1603 ± 2120	0.148	0.10
	After	2	5148 ± 3066	13314 ± 8977	8166 ± 5911		
Chironomid abundance (number·m^{-2})	Before	2	1414 ± 1477	1340 ± 1227	-75 ± 251	0.126	0.09
	After	2	1027 ± 356	2520 ± 1604	1493 ± 1248		
EPT abundance (number·m^{-2})	Before	2	3691 ± 307	6018 ± 451	2327 ± 758	0.067	0.11
	After	2	3610 ± 2228	9075 ± 5878	5465 ± 3650		
Richness (number of	Before	2	23.8 ± 2.8	27.9 ± 0.2	4.1 ± 2.7	0.596	0.06

Metric	Time	N	UBC1	MBT1	Difference between means at MBT1 and UBC1	*p*	Post-hoc power of the test
genera)	After	2	23.8 ± 2.1	23.9 ± 6.5	0.1 ± 8.7		
Simpson's Diversity Index (1-D)	Before	2	0.87 ± 0.04	0.88 ± 0.01	0.01 ± 0.03	0.352	0.05
	After	2	0.80 ± 0.06	0.75 ± 0.12	-0.05 ± 0.06		

Table 6. Mean difference in value of benthic invertebrate metrics (±SD) between UBC1 and LBC2 before and after the upgrade. The *p* values >0.1 indicate no significant effect of change in operations on the given metric. Data from 2009 were included in the "after" data.

Metric	Time	N	UBC1	LBC2	Difference between means at LBC2 and UBC1	*p*	Post-hoc power of the test
Total abundance of animals (number·m^{-2})	Before	2	6670 ± 3421	12251 ± 644	5581 ± 4065	0.666	0.07 to 0.1
	After	3	5706 ± 2373	14538 ± 6938	8833 ± 5068		
Chironomid abundance (number·m^{-2})	Before	2	1414 ± 1477	6151 ± 1081	4736 ± 396	0.877	0.05 to 0.06
	After	3	1170 ± 353	7860 ± 5945	6690 ± 5617		
EPT abundance (number·m^{-2})	Before	2	3691 ± 307	4419 ± 1461	728 ± 1768	0.736	0.06 to 0.07
	After	3	3778 ± 1602	4801 ± 1146	1023 ± 993		
Richness (number of genera)	Before	2	23.8 ± 2.8	27.5 ± 2.1	3.8 ± 0.7	0.505	0.07 to 0.08
	After	3	23.1 ± 1.9	26.2 ± 1.0	3.1 ± 1.3		
Simpson's Diversity Index (1-D)	Before	2	0.87 ± 0.04	0.71 ± 0.10	-0.16 ± 0.06	0.505	0.08 to 0.11
	After	3	0.79 ± 0.04	0.69 ± 0.13	-0.10 ± 0.12		

3.3.3 Multivariate tests of effect of change in flow on invertebrate assemblages

A frequency distribution showed many taxa had very high *D* values (many well over 10 and some approaching 1,000). This result justified the application of dispersion weighting to reduce the influence of taxa having the greatest degree of clumping.

There was a weak upstream to downstream pattern among all years before and after the upgrade (RELATE, Rho = 0.315, *p*= 0.003). Figure 15 shows that the assemblage at MBT1 was either equally dissimilar to that at UBC1 and LBC2 (2006, 2009, 2010) or that it was more dissimilar to that at UBC1 than to LBC2 (2005 and 2012). In either case, the assemblages at UBC1 differed from those at the downstream sites in varying amounts among years.

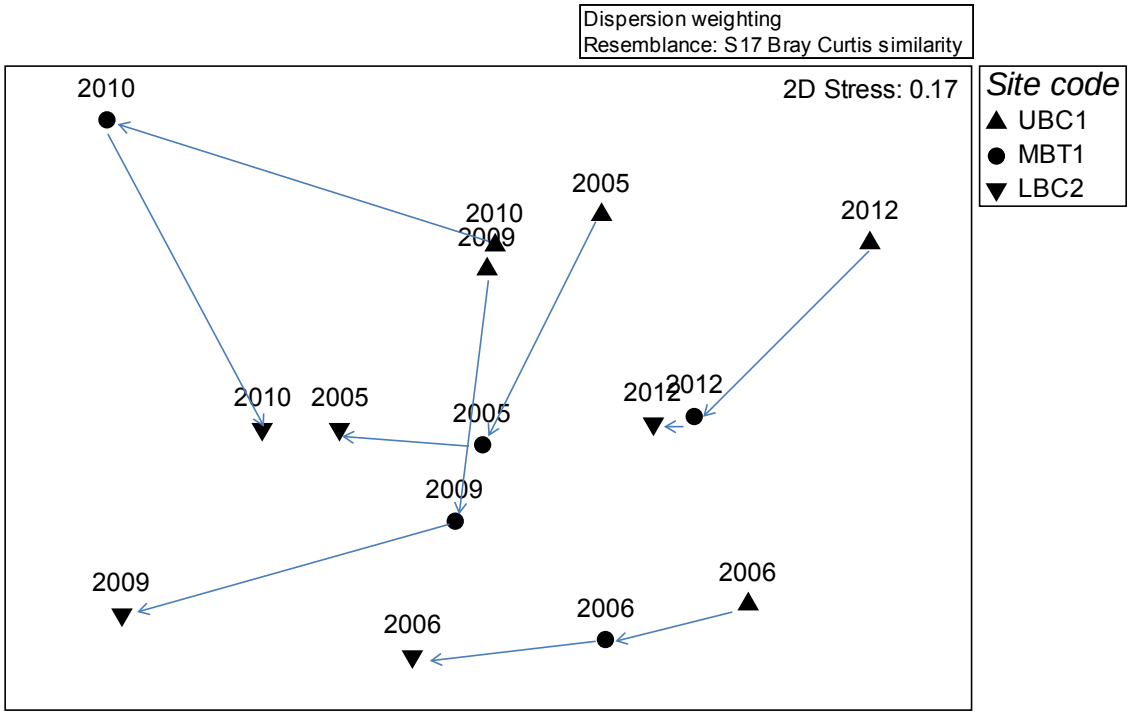


Figure 15. NMDS ordination of benthic invertebrate counts (identified to the most reliable taxonomic level down to genus) averaged by site and year. Years 2005, 2006 are before, and 2009, 2010, 2012 are after implementation of minimum flows. As a visual aide, arrows are drawn to highlight the upstream to downstream direction in each year.

R values from the two-way crossed ANOSIM using samples from UBC1 and MBT1 showed almost no site effect (R= 0.125, *p*=0.67) and a moderate time effect (R=0.5, *p*=0.11) (Table 7) on assemblages. R values for the ANOSIM using samples from UBC1 and LBC2 showed a moderate site effect (R=0.625 , *p*=0.22) and almost no time effect (R=0.125, *p*=0.44) (Table 7) on assemblages. These statistics do not lend

strong evidence of an effect of change in flow on assemblages at MBT1 or change in operations on assemblages at LBC2. A strong effect would show as R values exceeding 0.7 for both site and time.

Taxa contributing to these time and location effects were the EPT, chironomids, and naidid worms with a minor contribution from other taxa (Tables 8 and 9). Over half of the dissimilarities between times at UBC1 and MBT1 were cumulatively due to seven times more *Baetis* (mayflies) at MBT1 than at UBC1, a smaller increase in chironomids between UBC1 and MBT1 and a decline in abundance of the Capniidae (stoneflies) and Heptageniidae (mayflies) between UBC1 and MBT1. Only three taxa cumulatively contributed to over half of the dissimilarities between UBC1 and LBC2 at the different times (chironomids that were five times more abundant at LBC2 than at UBC1, and lower abundance of *Baetis* and naidid worms at LBC2 than at UBC1).

Table 7. Two-way crossed ANOSIM statistics to test location and time effects on dispersion weighted benthos counts for pairs of sites. The time effect represented blocks of years before (2005, 2006) and after (2010, 2012) the Aberfeldie dam upgrade.

Site Pair	Factor	Global R statistic	*p* value
UBC1 and MBT1	Site	0.125	0.67
	Time	0.5	0.11
UBC1 and LBC2	Site	0.625	0.22
	Time	0.125	0.44

Table 8. SIMPER output showing percent contribution of taxa cumulatively explaining 90% of dissimilarities of assemblages between UBC1 and MBT1 before and after the upgrade. The metric to which each taxon is a part is shown for reference.

Genus or other classification	**Metric**	**Mean abundance before the upgrade at UBC1 and MBT1 (number·m^{-2})**	**Mean abundance after the upgrade at UBC1 and MBT1 (number·m^{-2})**	**Percent contribution to dissimilarity between the before and after time periods at UBC1 and MBT1**	**Mean abundance at UBC1 at both times (number·m^{-2})**	**Mean abundance at MBT1 at both times (number·m^{-2})**	**Percent contribution to dissimilarity between UBC1 and MBT1 at both times**
Baetis	EPT	525	4028	28.2	1227	3326	25.6
Chironomids	Chironomids	1377	1774	12.7	1221	1930	12.8
Capniidae- young instars	EPT	798	651	7.7	394	1055	10.7
Heptageniidae-young instars	EPT	867	51	7.6	192	727	6.8
Naididae	Total	718	215	7.4	623	309	6.9
Baetidae-young instars	EPT	293	16	3.9	148	161	2.0
Sweltsa	EPT	277	24	3.4	244	58	2.3
Zapada	EPT	192	403	2.7	161	434	3.7
Diphetor	EPT	280	0	2.6	17	264	3.5
Rhithrogena Spp.	EPT	409	199	2.5	395	214	2.0
Antocha	Total	78	345	2.2	7	416	3.4
Taenionema	EPT	233	11	2.1	50	194	1.9
Nemouridae - young instars	EPT	163	156	1.9	192	127	1.6
Torrenticolidae	Total	97	165	1.7	110	152	1.4
Ephemerella sp.	EPT	243	164	1.5	108	300	2.2
Doddsia	EPT	0	100	1.1	2	99	1.5
Cinygmula	EPT	138	97	1.1	86	148	1.3

Table 9. SIMPER output showing percent contribution of taxa cumulatively explaining 90% of dissimilarities of assemblages between UBC1 and LBC2 before and after the upgrade. The metric to which each taxon is a part is shown for reference.

Genus or other classification	**Metric**	**Mean abundance before the upgrade at UBC1 and LBC2 (number·m^{-2})**	**Mean abundance after the upgrade at UBC1 and LBC2 (number·m^{-2})**	**Percent contribution to dissimilarity between the before and after time periods**	**Mean abundance at UBC1 at both times (number·m^{-2})**	**Mean abundance at LBC2 at both times (number·m^{-2})**	**Percent contribution to dissimilarity between sites**
Chironomids	Chironomids	3783	4515	23.9	1268	7176	44.4
Baetis	EPT	392	1705	17.9	1500	859	9.5
Naididae	Total	748	391	9.1	649	418	4.8
Capniidae- young instars	EPT	774	522	7.2	327	918	7.2
Baetidae-young instars	EPT	436	4	5.2	118	236	1.6
Heptageniidae-young instars	EPT	454	191	3.9	162	431	2.2
Sweltsa	EPT	253	50	3.4	216	46	1.2
Torrenticolidae	Total	366	176	2.9	88	416	2.9
Ephemerella sp.	EPT	362	416	2.4	114	675	4.4
Rhithrogena Spp.	EPT	353	199	2.4	401	120	2.2
Trombidiformes	total	0	241	2.1	63	225	1.7
Zapada	EPT	91	166	1.9	163	109	1.2
Nemouridae - young instars	EPT	109	91	1.9	154	42	1.2
Diphetor	EPT	210	184	1.8	13	375	2.7
Ephemerellidae-young instars	EPT	18	112	1.0	9	140	1.3
Lebertia	total	153	86	0.9	No contribution to the top 90% of dissimilarities		
Antocha	Total	23	123	0.9	17	148	1.1
Taenionema	EPT	88	23	0.9	No contribution to the top 90% of dissimilarities		
Cinygmula	EPT	88	109	0.7	No contribution to the top 90% of dissimilarities		

3.4 Periphyton

The NMDS ordination of periphyton assemblages showed clumping of samples by year, with assemblages from before the upgrade grouped together and those from after the upgrade separated apart from each other and from the pre-upgrade years (Figure 16).

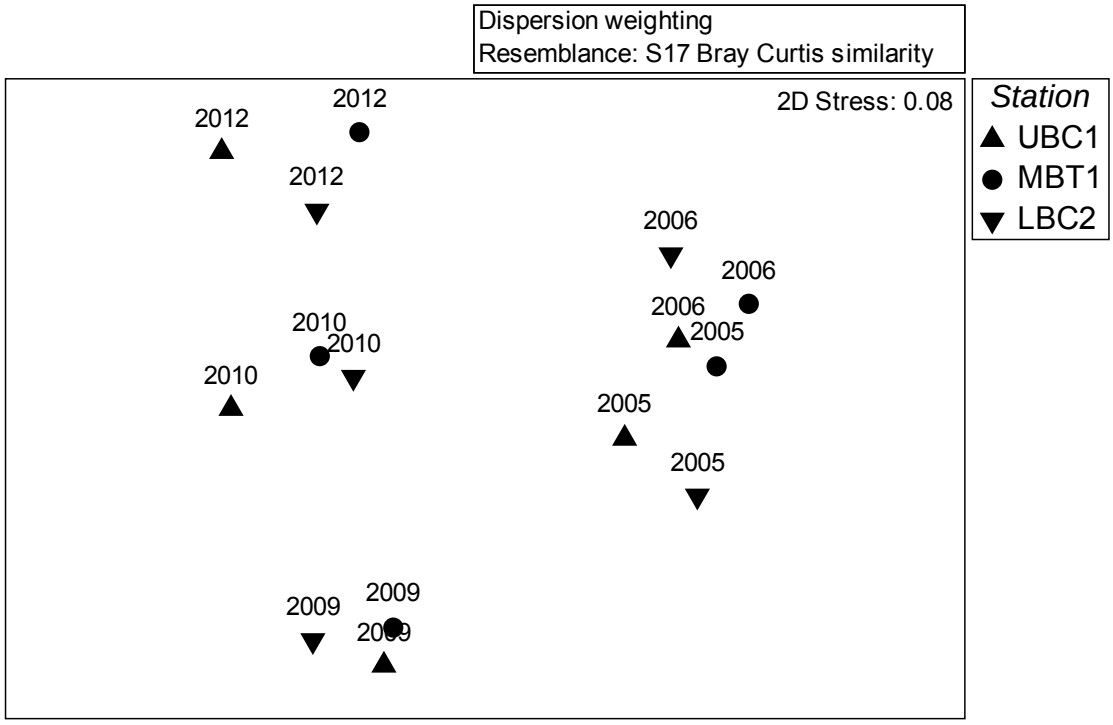


Figure 16. NMDS ordination of genus-level periphyton cell biovolume data averaged by site and year for main stem sites in the Bull River. Years 2005, 2005 are before, and 2009, 2010, 2012 are after implementation of minimum flows.

This temporal effect on assemblages was related to an increase in the biovolume of diatoms at UBC1 between the before and after time periods (Table 10). Changes of mean biovolume at the other sites and overlap of standard deviations showed the temporal changes at those sites were not significant. The diatoms were the single largest algal division in the Bull River accounting for >95% of total algal biovolume at all stations in both time periods. A total of 31 diatom genera were found with those accounting for most biovolume including *Diatoma sp*., *Achnanthidium sp*., *Fragilaria sp*., *Encyonema sp*., *Gomphonema sp*., *Rossithidium sp*., *Cymbella sp*., and *Didymosphenia sp.* Another factor contributing to the time effect (Figure 16) was the occurrence of several algal divisions after the upgrade that were not found before the upgrade, which resulted in increasing algal species richness over time (Table 10). Species heterogeneity was less affected and was high at all sites and times (Table 10). The new taxa found in later years were from the Chlorophyta, Cryptophyta, Cyanophyta (only at UBC1 and

LBC2), euglenoids, and dinoflagellates. These changes occurred at UBC1 as well as at the downstream sites, which means it was a change unrelated to the upgrade.

Table 10. Mean periphytic algal cell biovolume (±SD) by Division measured on artificial substrata at three sites in the Bull River before (2005 and 2006) and after (2010 and 2012) the Aberfeldie dam upgrade.

Algal division or metric	Time	N	Algal cell biovolume ($\mu m^3 \times 10^9/m^2$)		
			UBC1	MBT1	LBC2
Diatoms	Before	2	2679 ± 560	3271 ± 1145	2938 ± 10
	After	2	3667 ± 164	2887 ± 1021	3605 ± 1109
Chlorophyta	Before	2	0	0	0
	After	2	15.4 ± 21.8	47.5 ± 47.5	29.3 ± 27.8
Cryptophyta	Before	2	0	0	0
	After	2	13.8 ± 19.5	37.5 ± 47	17 ± 24
Cyanophyta	Before	2	0	14.6 ± 20.6	0
	After	2	17.6 ± 4.2	24.3 ± 23.7	55.8 ± 70.9
Euglenoids	Before	2	0	0	0
	After	2	13.4 ± 19.0	18.3 ± 25.9	9.2 ± 12.9
Dinoflagellates	Before	2	0	0	0
	After	2	6.1 ± 8.6	22.9 ± 32.4	12.2 ± 17.3
All Divisions	Before	2	2679 ± 560	3286 ± 1124	2938 ± 10
	After	2	3733 ± 91	3025 ± 1197	3729 ± 1098
Richness (number of genera)	Before	2	9.2 ± 1.2	10.0 ± 3.3	9.2 ± 2.1
	After	2	17.3 ± 3.3	18.8 ± 2.8	21.0 ± 0.7
Simpson's Diversity Index (1-D)	Before	2	0.81 ± 0	0.8 ± 0.08	0.76 ± 0.03
	After	2	0.75 ± 0.05	0.87 ± 0.02	0.82 ± 0.03

Mean periphyton PB was consistently low, ranging between 4 mg chl-a·m^{-2} among all sites in 2006 and 36 mg chl-a·m^{-2} at LBC2 in 2009 (Figure 17). The t-test run on the BACI layout of PB measurements showed no significant effect of the change in flow on PB at MBT1 (p>0.1) (Table 11) and no significant effect of change in operations on PB at LBC2 (Table 12). Due to the low number of replicates (years) in the BACI layout, the power of these tests was weak, ranging from 0.13 for the test of effect of the

change in flow at MBT1 on PB (Table 11) to 0.39 for the test of effect of change in operations at LBC2 on PB (Table 12).

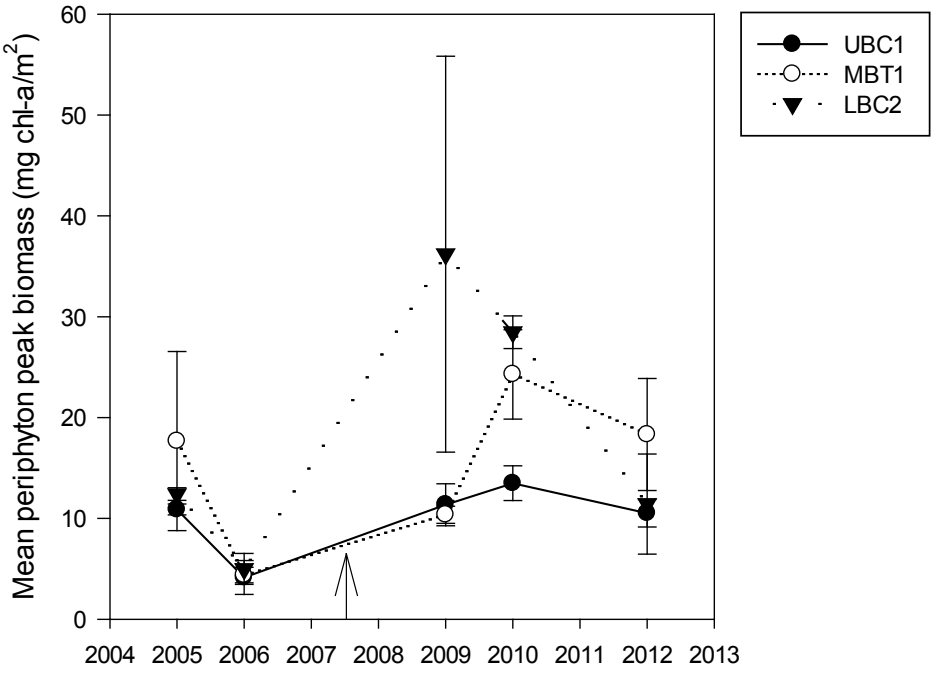


Figure 17 Mean periphyton PB (±SD) at UBC1, MBT1 and LBC2 before (2005 and 2006) and after the upgrade (2009, 2010, and 2012). Arrow points to entry in operations of new flow regime.

Table 11. Mean difference in PB (±SD) between UBC1 and MBT1 before and after the upgrade. *P* values >0.1 indicate no significant effect of change in flow on PB. Data from 2009 were not included in the "after" data because flow in that year greatly exceeded the minimum flow to be tested.

Metric	Time	N	UBC1	MBT1	Difference between means at MBT1 and UBC1	*P*	Post-hoc power of the test
PB (mg chl-a·m^{-2})	Before	2	7.5 ± 4.8	11.0 ± 9.4	3.5 ± 4.6	0.249	0.13
	After	2	12.0 ± 2.1	21.3 ± 4.2	9.3 ± 2.1		

Table 12. Mean difference in PB (±SD) between UBC1 and LBC2 before and after the upgrade. *P* values >0.1 indicate no significant effect of change in operations on PB. Data from 2009 were included in the "after" data because there was no change in flow at LBC2 related to operations.

Metric	Time	N	UBC1	LBC2	Difference between means at LBC2 and UBC1	*P*	Post-hoc power of the test
PB (mg chl-a·m^{-2})	Before	2	7.5 ± 4.8	8.7 ± 5.3	1.2 ± 0.5	0.439	0.39
	After	3	12.0 ± 2.1	19.9 ± 12.1	7.9 ± 9.9		

4 DISCUSSION

After the change in flow regime in 2009, flows in August and September of each year were variable but they did meet recommendations from the Water Use Planning Consultative Committee and Fisheries Technical Committee (BC Hydro 2006). Flows did not decline below the minimum level of 2 $m^3 \cdot s^{-1}$ during the study period but they were not always at this minimum rate. The maintenance outage at the powerhouse in 2009 caused all flow in the Bull River to pass through the diversion reach. Benthic invertebrate data from that year were removed from testing the effect of the upgrade on various biological metrics to avoid confounding by this anomalous flow that was not part of normal operating procedures. In the other years after the upgrade (2010 and 2012), short term and small increases in flow above 2 $m^3 \cdot s^{-1}$ did occur in the diversion reach due to occasional spills as noted in Section 3.1. These events were considered part of normal operating conditions.

Benthic invertebrate communities upstream and downstream of the Aberfeldie Dam were diverse and abundant. Richness (20 to 28 genera or higher taxa per sample), sample heterogeneity (Simpson's Index; 0.7 to 0.9), and average invertebrate density among sites and times (5,100 – 14,500 animals·m^{-2}) were all within the range found in other mountain streams in British Columbia including the Cheakamus River (Perrin 2010), the lower Capilano River (Perrin 2004), and the lower Coquitlam River (Perrin and Bennett 2011). Deegan et al. (1997) reported aquatic insect densities of 5,000 – 15,000 animals·m^{-2} in the Kuparuk River, Alaska. Wipfli et al. (1998) reported densities of 1,000 – 11,000 animals·m^{-2} in another Alaskan stream. These densities increased to 40,000 animals·m^{-2} in the presence of decomposing salmon carcasses. Rosario and Resh (2000) reported densities up to 18,000 animals·m^{-2} and an average of 13,761 animals·m^{-2} in perennial streams of northern California. Densities of 1,500 – 40,000 animals·m^{-2} were reported by Dewson et al. (2007a) among several undisturbed streams

in New Zealand. Densities of 6600 – 14,000 animals·m^{-2} were reported by Rader and Belish (1999) among undisturbed streams of the Rocky Mountains in Colorado. All these comparisons show that invertebrate densities in the Bull River were within a range expected in an undisturbed mountain river.

Invertebrate densities of 5100 – 13000 animals·m^{-2} in the diversion reach at MBT1 may be overestimates of actual densities over the whole reach. That segment was characterized by large cobble and boulder, interspersed by smaller cobble, gravel, and smaller pockets of sand. This relatively small material could be sampled within the 900 cm^2 Surber sampler but the larger boulders could not. Relatively few invertebrates would be expected to inhabit the face of boulders compared to abundance found in the diversity of interstitial spaces between smaller substrates. Hence, densities in the Surber samples were potentially higher than would be found associated with the boulders and are expected to be maximum values among all substrata of the whole segment. Since cobble and gravel is optimum for hosting benthic invertebrates, the data are considered representative of the best substrates for supporting benthic invertebrates in the test reach.

Because MBT1 was located immediately downstream of the diversion reach canyon, which may not be suitable to support invertebrates, most of the invertebrate recruitment to the test reach may have originated either from the headpond, from reaches upstream of the headpond, from egg deposition by adults directly in the test reach, or by some combination of these sources. The headpond was morphologically and hydrologically similar to a small shallow lake or a slowing of the river. It would be expected to be well oxygenated, given a rapid flushing rate, and with a long history of infilling with sediment of inorganic and organic origin that is now at a level close to the dam crest (Cope 2005), the headpond would be expected to produce a wide assemblage of benthic invertebrates.

In contrast, invertebrate assemblages upstream and downstream of dams that impound reservoirs having deep pelagic zones, small littoral zones that are seasonally dewatered, and water residence times lasting weeks to years can be different (Takao et al. 2008, Katano et al. 2009). Lentic environments will not support a diverse assemblage of insects that are adapted to water flowing over stony substrates. Some similarity among invertebrate assemblages between UBC1 (upstream of the headpond) and MBT1 (downstream of the headpond) in all years shows that a break in connectivity caused by the headpond was not present in the Bull River.

The increase in EPT abundance with lower median flows at MBT1 is consistent with numerous studies showing various sensitivities of benthos to flow (e.g., Gore et al. 2001, Merigoux and Doledec 2004, and Nelson and Lieberman 2002). Genera contributing most to this flow effect included *Baetis, Capniidae, Heptageniidae, Zapada, Diphetor, Taenionema, Ephemerella, Dodsia*, and *Cinygmula*, with *Baetis* contributing the most. In a review of worldwide studies, Poff and Zimmerman (2010) found that the

direction of response by biological communities to alteration of flow is not always the same among rivers. Similarly, Dewson et al. (2007a) showed that abundance of benthic invertebrates can increase or decrease in response to decreased flow. Using experimental water diversions, Dewson et al. (2007b) further showed a decline in the proportion of EPT and a decline of total density of invertebrates in response to artificially decreased flow in a pristine mountain stream in New Zealand. In a diversion experiment in Michigan, Wills et al (2006) found density of EPT declined by several fold when flow was lowered by 90% but less of a response when flows were dropped by 50%. The same direction of response and extent of response to different reductions in flow were reported in earlier studies by Rader and Belish (1999). In contrast, Cobb et al. (1992) and Acuna et al. (2005) found an inverse correlation between flow and benthic invertebrate density. Links of biota to flow can be stronger than links to other habitat attributes (Armanini et al. 2010), but those other attributes can modify associations between benthos and flow (Perrin 2010, Perrin and Bennett 2011) and potentially contribute to variation among hydraulic – biotic links.

Despite this confounding by various habitat attributes, a response of EPT to change in flow in the diversion reach was found in the Bull River. It may be related to tolerances of the EPT to change in hydraulic stress (e.g., Rempel et al. 1999) at the different flows. Highest densities occurred at low flows when water depths and velocities would be lower than during high flows. Other studies have shown that this kind of ordering of communities is related to sensitivity of benthos to hydraulic gradients with many occurring in highest density in shallow water where hydraulic stress is lowest (Rempel et al. 2000). A shift to lower flows in August and September after the upgrade at Aberfeldie may have caused hydraulic stress to decline, potentially favouring EPTs.

The periphyton communities were comprised mainly of diatoms, which is typical among streams and rivers of all sizes in British Columbia. The diatoms are ubiquitous and they are a primary food source for grazing and collecting invertebrates that are common in the Bull River (Table 13).

Table 13. Feeding attributes of benthic invertebrates found in the Bull River.

Genus or other classification	Metric	Functional feeding group*	Food source**
Baetis	EPT	Scrapers, collector - gatherers	Periphyton and detritus
Chironomids	Chironomids	Collector – gatherers	Detritus
Capniidae	EPT	Shredders	Leaf matter and detritus
Heptageniidae	EPT	Scrapers, collector - gatherers	Periphyton and detritus
Naididae	Total (belongs to the subclass Oligochaeta)	Collector – gatherers	Detritus
Sweltsa	EPT	Predators of chironomids	Chironomids
Zapada	EPT	Shredders and detritivores	Leaf matter and detritus
Diphetor	EPT	Collector - gatherer	Detritus
Rhithrogena	EPT	Collector – gatherers,	Detritus and periphyton

Genus or other classification	Metric	Functional feeding group*	Food source**
		scrapers	
Antocha	Total (belongs to the order Diptera)	Collector - gatherer	Detritus
Taenionema	EPT	Scrapers	Periphyton
Nemouridae	EPT	Shredders, collector - gatherers	Leaf matter and detritus
Torrenticolidae	Total (belongs to the subclass Acari)	Predators	Other invertebrates
Ephemerella	EPT	Collector – gatherers	Detritus and periphyton
Doddsia	EPT	Scraper	Periphyton
Cinygmula	EPT	Scrapers, collector - gatherers	Diatoms and detritus
Trombidiformes	Total (belongs to the subclass Acari)	Not known	Not known
Lebertia	Total (belongs to the subclass Acari)	Predator	Other invertebrates

*from Merritt and Cummins (1996) and Barbour et al. (1999)
**Food source is implied from the functional feeding group where scrapers feed on periphyton (algae, fungi, bacteria on the surface of stones), collector – gatherers feed on detrital matter associated with senescent algae and decomposing leaf and other organic matter of riparian origin, shredders feed on large leaf and other organic matter, and predators feed on other animals of a size they can ingest that are usually invertebrates (Merritt and Cummins, 1996).

The increased richness of algal taxa after the upgrade compared to before the upgrade cannot be attributed to changes at Aberfeldie because it occurred at all sites, including the control site at UBC1. There was a change in taxonomists between the before and after years which may explain some of the differences. The “new” taxa in the after years (Chlorophyta, Cryptophyta, euglenoids, dinoflagellates) may actually have been present in the “before” years but were missed in the algal identifications. Alternatively, a regional change may be occurring that favours a more diverse assemblage of benthic algae. The habitat data in Table 4 shows that soluble reactive phosphorus concentration was the only habitat attribute that changed over time although the differences may not be as much as the data shows given challenges to resolve differences at the low concentrations found in the Bull River. If real, the increase in soluble reactive phosphorus concentration shown in Table 4 may have favoured the increase in diversity of algal taxa. Other attributes had values much the same between the two blocks of years.

Average PB on the artificial substrata among sites and years (4 - 36 mg chl-a·m^{-2}) was within a range that is considered typical of moderately enriched streams (3 – 60 mg chl-a·m^{-2}; Biggs 1996). PB at MBT1 was not affected by change in flow and PB at LBC2 was not affected by the change in operations. The t-tests for distinguishing these effects, however, had very low power which means the tests had little chance of detecting a flow or operations effect if it was present. Hence, the doubling of PB at MBT1 between the before and after years compared to the 60% increase at UBC1 cannot be ignored but also cannot be considered conclusive evidence of change. The same

argument can be made for the increase in PB at LBC2 over time. One factor potentially contributing to PB over time among all sites was the concentration of soluble reactive phosphorus that approximately tripled between the before and after years. While soluble reactive phosphorus concentration was always low, the changes over time were sufficiently different to produce the differences in PB, based on P-limited accrual curves reported by Bothwell (1989). These curves show that soluble reactive phosphorus concentrations <1 $\mu g \cdot L^{-1}$ can limit cellular growth rates (Bull River before the Aberfeldie upgrade) but concentrations exceeding 1 $\mu g \cdot L^{-1}$ can saturate growth rates (Bull River after the upgrade) and lead to high areal biomass. Other factors that can influence the accrual of periphyton including temperature (Bothwell 1988), turbidity and its effect on irradiance that can influence photosynthetic rates (Hill 1996), and particle size distribution (Burkholder 1996) did not change temporally and thus would not contribute to the change in PB. The lower flows at MBT1 after the upgrade potentially enhanced any effect of higher soluble reactive phosphorus concentrations by many of the same processes affecting the benthic invertebrates including lower water velocities that would diminish drag forces but still maintain steep nutrient concentration gradients around cells that are needed to optimize cellular growth rates in the algal mats (Stevenson 1996). Little variance of flow in the absence of spill may also favour high areal biomass of algae, as found in stable flows downstream of dams in New Zealand (Tonkin et al. 2009). These low flow conditions could reduce sloughing and maintain high algal growth rates according to nutrient supply.

As was found for algal PB, low power of the statistical tests showed the tests had little chance of detecting a flow or operations effect if it was present. Power of the test on EPT in the test segment was not calculated nor relevant because a flow effect on that metric was found. Mean abundance of all invertebrates and chironomids in the test reach did increase between the before and after years by 60% and 88% respectively. These changes are consistent with the significant increase in EPT at lower flows and generally support evidence of benthos response to hydraulic gradients that was described above. The supplementary multivariate tests of time and location effects were not, however, supportive of a flow effect on benthos. The important R values were low for both time and location effects on invertebrate assemblages. This result in combination with lack of an effect of change in flow or change in operations on invertebrate metrics leads to the conclusion that the change in flow did not significantly change the abundance of the total assemblage of invertebrates and chironomids that were a large part of the total assemblage. Within that total assemblage, EPT abundance did increase after lower flows were implemented in the diversion reach.

One of the attributes that can modify associations between EPT density and flow is algal biomass that provides food for invertebrates (McCutchan and Lewis 2002). If PB increased after the upgrade, it may have supported greater densities of the EPT by increasing availability of food, mainly for the grazers and collector – gatherers (Table 13). The extent of this food web interaction cannot be resolved with existing data but its

potential importance in modifying links between benthos and flow supports well known evidence of trophic interactions in streams (Feminella and Hawkins 1995, Rosemond et al. 2000, Perrin and Richardson 1997).

In summary, management questions 1 and 2 (Section 1) have been answered. In data compiled to date, there was no significant effect of post redevelopment change in flow on the composition, diversity and biomass of periphyton. Also in the data compiled to date we found no significant effect of the post redevelopment change in flow on the composition, diversity and biomass of total benthic invertebrates. Within the invertebrate assemblage, the EPT that are important fish food organisms significantly increased as a result of the post redevelopment change in flow. The results showing no effect of the post redevelopment change in flow on periphyton and invertebrate metrics had low power due to small sample size, where sampled size is the number of years of observations. Additional sampling in future years would be required to improve power of the analysis and increase confidence in test results.

5 LITERATURE CITED

Acuna, V., I. Munoz, A. Giogi, M. Omella, F. Sabater, and S. Sabater. 2005. Drought and postdrought recovery cycles in an intermittent Mediterranean stream: structural and functional aspects. Journal of North American Benthological Society. 24(4): 919-933.

Armanini, D.G., N. Horrigan, W.A. Monk, D.L. Peters, and D.J. Baird. 2010. Development of a benthic macroinvertebrate flow sensitivity index for Canadian rivers. River Research and Applications. www.interscience.wiley.com DOI: 10.1002/rra.1389.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C. Available at: http://water.epa.gov/scitech/monitoring/rsl/bioassessment/app_a.cfm .

BC Hydro. 2009. BC Hydro completes new Aberfeldie generating station. BC Hydro, Burnaby, B.C. Available at http://www.bchydro.com/news/press_centre/news_releases/2009/bchydro_completes_new_aberfeldie_generating_station.html. Viewed April, 2013.

BC Hydro. 2008. Aberfeldie project water use plan: Monitoring program terms of reference, ABFMON-2 primary and secondary productivity monitoring. BC Hydro, Burnaby, B.C.

BC Hydro. 2006. Aberfeldie redevelopment project water use plan. BC Hydro, Burnaby, B.C. Available at http://www.bchydro.com/content/dam/hydro/medialib/internet/documents/environment/pdf/wup_water_use_plan_aberfeldie_redevelopment_project.pdf

BC Hydro. 2004. Map of general arrangement of existing plant. Aberfeldie redevelopment project. BC Hydro, Burnaby, B.C.

Bothwell, M.L. 1989. Phosphorus-limited growth dynamics of lotic periphyton diatom communities: areal biomass and cellular growth rate responses. Can. J. Fish. Aquat. Sci. 46:1293-1301.

Bothwell, M.L. 1988. Growth rate responses of lotic periphytic diatoms to experimental phosphorus enrichment: The influence of temperature and light. Canadian Journal of Fisheries and Aquatic Sciences. 45: 261-270.

Boulton, A.J. 1999. An overview of river health assessment: philosophies, practice, problems and prognosis. Freshw. Biol. 41: 469-479.

Burkholder, J.M. 1996. Interactions of benthic algae with their substrata. In: R.J. Stevenson, M.L. Bothwell and R.L Lowe (eds). Algal Ecology. Academic Press. San Diego.

Cavanagh, N., R.N. Nordin, and P.D. Warrington. 1997. Freshwater Biological Sampling Manual. BC Ministry of Environment Lands and Parks. 41p

Clarke, K.R. and R.N. Gorley. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E: Plymouth. UK.

Cobb, D.G., T.D. Galloway and J.F. Flannagan. 1992. Effects of discharge and substrate stability on density and species composition of stream insects. Canadian Journal of Fisheries and Aquatic Sciences. 49: 1788-1795.

Cope, S. 2005. Environmental impact assessment of the Aberfeldie Dam redevelopment project, Bull River, British Columbia. Report prepared by Westslope Fisheries Ltd. for BC Hydro, Burnaby, BC.

Deegan, L.A., B.J. Peterson, H. Golden, C.C. McIvor, and M.C. Miller. 1997. Effects of fish density and river fertilization on algal standing stocks, invertebrate communities, and fish production in arctic tundra river. Can. J. Fish. Aquat. Sci. 54: 269-283.

Demarchi, D.A., R.D. Marsh, A.P. Harcombe, and E.C. Lea. 1990. The Environment. p.55 – 144. In: R.W. Campbell, N.K. Dawe, I. McTaggart-Cowan, J.M. Cooper, G.W. Gaiser, and M.C.E. McNall. (Ed.). The Birds of British Columbia. UBC Press. Vancouver.

Dewson, Z.S., A.B.W. James, and R.G. Death. 2007a. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. Journal of North American Benthological Society 26(3): 401-415.

Dewson, Z.S., A.B.W. James, and R.G. Death. 2007b. Invertebrate community responses to experimentally reduced discharge in small streams of different water quality. Journal of the North American Benthological Society. 26: 754-766.

Edmondson, W.T. 1959. Fresh water biology. 2nd ed. John Wiley & Sons, New York. 1248 pp.

Environment Canada 2012. Metal Mining Environmental Effects Monitoring (EEM) Technical Guidance Document. Available at http://www.ec.gc.ca/Publications/ Accessed March 2013.

Feminella, J. W. and C.P. Hawkins. 1995. Interactions between stream herbivores and periphyton: a quantitative analysis of past experiments. Journal of the North American Benthological Society. 14: 465-509.

Gore, J.A., J.B. Layzer, and J. Mead. 2001. Macroinvertebrate instream flow studies after 20 years: A role in stream management and restoration. Regulated Rivers: Research and Management. 17: 527-542.

Guildford, S.J. and R.E. Hecky. 2000. Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship? *Limnology and Oceanography* 45: 1213-1223.

Healey, F.P. 1985. Interacting effects of light and nutrient limitation on the growth rate of *Synechococcus linearis* (Cyanophyceae). *Journal of Phycology* 21: 134-146.

Hill, W.R. 1996. Effects of light. In: R.J. Stevenson, M.L. Bothwell and R.L Lowe (eds). Algal Ecology. Academic Press. San Diego.

Holm-Hansen, O., C.J. Lorenzen, R.W. Holmes, and J.D.H. Strickland. 1965. Fluorometric determination of chlorophyll. J. Cons. Perm. Int. Explor. Mer. 30:3-15.

Hurlbert, S.J. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs. 54: 187-211.

Katano, I., J. Negishi, T. Minagawa, H. Doi, Y. Kawaguchi, and Y. Kayaba. 2009. Longitudinal macroinvertebrate organization over contrasting discontinuities: effects of a dam and a tributary. Journal of the North American Benthological Society. 28: 331-351.

Krebs, C.J. 1999. *Ecological Methodology*. Addison Wesley. Longman, NY.

McCutchan, J.H. Jr. and W.M. Lewis Jr. 2002. Relative importance of carbon sources for macroinvertebrates in a Rocky Mountain stream. Limnology and Oceanography 47: 742-752.

McPhail, J.D. and P.M. Troffe. 1998. The mountain whitefish (*Prosopium williamsoni*): a potential indicator species for the Fraser System. DOE FRAP 1998-16 report prepared for Environment Canada, Vancouver BC.

McPherson, S., M. Robinson and C. Good. 2010. Aberfeldie Sidechannel – Effectiveness Monitoring of Fish Habitat Works. Prepared for British Columbia Hydro and Power Authority, Burnaby, BC. Prepared by Interior Reforestation Co. Ltd., Cranbrook, BC. 74 pp + appendices.

Menzel, D.W. and N. Corwin. 1965. The measurement of total phosphorus in seawater based on liberation of organically bound fractions by persulfate oxidation. Limnology and Oceanography. 10: 280-282.

Merigoux, S., and S. Doledec. 2004. Hydraulic requirements of stream communities: a case study on invertebrates. Freshwater Biology 49: 600-613.

Merritt, R.W. and K.W. Cummins, eds. 1996. An introduction to the aquatic insects of North America, 3rd ed. Kendall/Hunt, Dubuque, Iowa. 862 pp.

Merritt, R.W., V.H. Resh, and K.W. Cummins. 1996. Design of aquatic insect studies: Collecting, sampling and rearing procedures. p12 – 28. In: R.W. Merritt and K.W. Cummins (Ed.). An Introduction to Aquatic Insects of North America. Kendall Hunt Publ. Dubuque Iowa. 862p.

Ministry of Environment, Lands and Parks. 1988. Guidelines for Interpreting Water Quality Data. Prepared for the Land Use Task Force Resources Inventory Committee Version 1.0. Available at: http://archive.ilmb.gov.bc.ca/risc/pubs/aquatic/interp/index.htm. Accessed October 2012.

Murphy, J.A. and J.P. Riley. 1962. A modified single solution method for the determination of inorganic phosphate in natural waters. Anal. Chim. Acta. 27: 31-36.

Nelson, S.M., and D.M. Lieberman. 2002. The influence of flow and other environmental factors on benthic invertebrates in the Sacramento River, U.S.A. Hydrobiologia 489: 117-129.

Norris, R.H. and C.P. Hawkins. 2000. Monitoring river health. Hydrobiologia 435: 5-17.

Norris, R.H. and M.C. Thoms. 1999. What is river health? Freshwater Biology 41: 197-209.

Nusch, E.A. 1980. A comparison of different methods for chlorophyll and phaeopigment analysis. Arch. Hydrobiol. Beih. Ergebn. Limnol. 14: 14-36.

Pennak, R.W. 1978. Fresh water invertebrates of the United States, 2nd ed. John Wiley & Sons, New York. 803 pp.

Perrin, C.J. 2010. Cheakamus River benthic community monitoring for water license requirements: Cheakamus River monitoring program #7. Report prepared by Limnotek Research and Development Inc. for BC Hydro. 59p.

Perrin, C.J. 2004. Effect of drawdown and construction in the Capilano Reservoir on periphyton and benthic invertebrate communities in the lower Capilano River. Report prepared by Limnotek Research and Development Inc. for B.C. Ministry of Water Land and Air Protection. 62p.

Perrin, C.J. and S. A. Bennett. 2013b. Benthic assemblages in a constructed side channel as compensation for water diversion in the Bull River, British Columbia: BC Hydro project number ABFMON#5-2A. Report prepared by Limnotek Research and Development Inc. for BC Hydro. 45p.

Perrin, C.J. and S.A. Bennett. 2011. Coquitlam River periphyton and benthic invertebrate monitoring: Coquitlam River Monitoring Program #5. Prepared by Limnotek Research and Development Inc. for BC Hydro. 73p.

Perrin, C.J., M.L. Bothwell and P.A. Slaney. 1987. Experimental enrichment of a coastal stream in British Columbia: effects of organic and inorganic additions on autotrophic periphyton production. Canadian Journal of Fisheries and Aquatic Sciences. 44: 1247-1256.

Perrin, C.J. and C. Canning. 2010. Change in wetted habitat area due to water diversion in the Bull River, British Columbia: Final report for project ABFMON#3. Report prepared by Limnotek Research and Development Inc. for BC Hydro. 16p.

Perrin, C.J. and J.S. Richardson. 1997. N and P limitation of benthos abundance in the Nechako River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences. 54: 2574-2583.

Poff, N.L. and J.K.H. Zimmerman. 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 55: 194-205.

Rader, R.B. and T.A. Belish. 1999. Influence of mild to severe flow alterations on invertebrates in three mountain streams. Regulated Rivers Research and Management. 15: 353-363.

Reice, S.R. and M. Wohlenberg. 1993. Monitoring freshwater benthic macroinvertebrates and benthic processes: measures for assessment of ecosystem health. In: Rosenberg, D.M. and V.H. Resh. (Ed). 1993. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall. New York.

Rempel, L.L., J.S. Richardson, and M.C. Healey. 2000. Macroinvertebrate community structure along gradients of hydraulic and sedimentary conditions in a large gravel-bed river. Freshwater Biology 45: 57-73.

Rempel, L.L., J.S. Richardson, and M.C. Healey. 1999. Flow refugia for benthic macroinvertebrates during flooding of a large river. J. N. Am. Benthol. Soc. 18: 34-48.

Rhee, G.Y. 1978. Effects of nitrogen: Phosphorus atomic ratios and nitrate limitation on algal growth, cell composition, and nitrate uptake. *Limnology and Oceanography* 23: 10-25.

Rhee, G.Y. and I.J. Gotham. 1980. Optimum nitrogen: Phosphorus ratios and coexistence of planktonic algae. *Journal of Phycology* 16: 486-489.

Rosario, R.B., and V.H. Resh. 2000. Invertebrates in intermittent and perennial streams: is the hyporheic zone a refuge from drying? Journal of the North American Benthological Society 19(4): 680-696.

Rosemond, A.D., P.J. Mulholland, and S.H. Brawley. 2000. Seasonally shifting limitation of stream periphyton: response of algal populations and assemblage biomass and productivity to variation in light, nutrients, and herbivores. Canadian Journal of Fisheries and Aquatic Sciences. 57: 66-75.

Schoby, G.P. and E.R. Keeley. 2011. Home range size and foraging ecology of Bull Trout and Westslope Cuttroat Trout in the Upper Salmon River Basin, Idaho. Transactions of the American Fisheries Society 140(3): 636-645.

Stainton, M.P., M.J. Chapel, and F.A. Armstrong. 1977. The chemical analysis of freshwater. 2nd. ed. Fisheries and Environment Canada Miscellaneous Special Publications. 25.

Stevenson, R.J. 1996. The stimulation and drag of current. In: R.J. Stevenson, M.L. Bothwell and R.L Lowe (eds). Algal Ecology. Academic Press. San Diego.

Stewart-Oaten, A., W.W. Murdoch and K.R. Parker. 1986. Environmental impact assessment: "pseudoreplication" in time?". Ecology 67(4): 929-940.

SYSTAT Software, Inc. 2004. SYSTAT 11 Statistics 1. Richmond, CA.

Takao, A., Y. Kawaguchi, T. Minagawa, Y. Kayaba, and Y Morimoto. 2008. The relationships between benthic macroinvertebrates and biotic and abiotic environmental characteristics downstream of the Yahagi Dam, central Japan, and the state change caused by inflow from a tributary. River Research and Applications. 24: 580-597.

Taylor, B.W., A.R. McIntosh, and B.L. Peckarsky. 2002. Reach-scale manipulations show invertebrate grazers depress algal resources in streams. Limnology and Oceanography. 47: 893-899.

Tonkin, J.D., R.G. Death and M.K. Joy. 2009. Invertebrate drift patterns in a regulated river: dams, periphyton biomass or longitudinal patterns? Rivers Research and Applications 25: 1219-1231.

Wetzel, R.G. 2001. *Limnology.* Academic Press. San Diego.

Wills, T.C., E.A. Baker, A.J. Nuhfer and T.G. Zorn. 2006. Response of the benthic macroinvertebrate community in a northern Michigan stream to reduced summer streamflows. River Research and Applications 22: 819-836.

Wipfli, M.S., J. Hudson, and J. Caouette. 1998. Influence of salmon carcasses on stream productivity: response of biofilm and benthic macroinvertebrates in southeastern Alaska, USA. Can. J. Fish. Aquat. Sci. 55: 1503-1511.

Wolman. M. 1954. A method of sampling coarse bed material. American Geophysical Union Transactions. 35:951-956.