Aberfeldie Water Use Plan

EFFECTIVENESS MONITORING FISH HABITAT WORKS,TASK 2A: PRIMARY AND SECONDARY PRODUCTIVITY MONITORING

Reference: ABFMON#5-2A

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Study Period: 2009 to 2012

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Cover photo: View of a riffle in the Aberfeldie constructed side channel, 2009. Photo by Chris Perrin.

EXECUTIVE SUMMARY

Change in capacity of a diversion reach to produce benthic invertebrates that are food for fish was measured following a 2009 upgrade to the Aberfeldie hydropower project. Productive capacity of a constructed side channel was also examined. Calculations were run to determine if the diversion reach and side channel following the upgrade attained no net loss of productive capacity. Before the upgrade, water flowed over the Aberfeldie dam spillway during eight to nine months from late March to late December annually 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, all flow in the river is diverted through the penstock except for a minimum flow release to the diversion reach between the dam and powerhouse that was negotiated as part of water use planning. 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 combined abundance 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 and are the main prey of fishes in the Bull River increased by 23% within the wetted area of the diversion reach due to the reduced flow. This change was the net effect of a 50% increase in density and a 19% decline in wetted area. There was a 19% decline in abundance of invertebrates among all taxa and chironomids within that assemblage, entirely due to the decline in wetted area. Abundance in the side channel was sufficient to offset that loss, resulting in no net change of benthic invertebrate productive capacity.

Consideration was given to potential effect of releasing flows lower than 2 $m^3 \cdot s^{-1}$ in the summertime on the abundance of benthic invertebrates. The effect of lower flows on benthic assemblages could not be tested because there was no flow release lower than 2 $m^3 \cdot s^{-1}$ over the course of the study. Review of literature showed no predictable direction of change. Given the importance of the EPT as food for fish in the Bull River, risk of a sudden drop in abundance of the EPT at diminished flow was considered high and could lead to a net loss of fish food organisms. The side channel is not capable of offsetting potential loss of the EPT and other invertebrates ingested by fish in the diversion reach because dissolved oxygen concentrations are lower than those required to support various life stages of aquatic organisms. To maintain production of fish food organisms in the Bull River, no further reduction in summertime flow to the diversion reach should be contemplated unless improvements are made to the side channel and these are shown to be effective.

Status of objectives, management questions, and hypotheses for ABFMON#5-2A

Project number	Objectives	Management questions	Management hypotheses	Status following work in 2012	Page(s) showing the result for ABFMON5-2a
ABFMON#5-2a	To determine if the combined productive capacity in the diversion reach of the Bull River and a constructed side channel at Aberfeldie attained no net loss of productive capacity following the Aberfeldie redevelopment project	If changes in the benthic community associated with post-redevelopment facility operations are detected (in ABFMON2), 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?	*Periphyton* Ho_1: There is no difference between the peak biomass of periphyton in the diversion reach before redevelopment and in the combined off channel habitat and diversion reach of Bull River under the post redevelopment 2 m^3/s minimum summer flow release. Ho_2: There is no difference between the total abundance and diversity of periphyton in the diversion reach before redevelopment and in the combined off channel habitat and diversion reach of Bull River under the post redevelopment 2 m^3/s minimum summer flow release. *Benthic invertebrates* Ho_3: There is no difference between the total abundance, biomass and diversity of benthic invertebrates in the diversion reach before redevelopment and in the combined off channel habitat and diversion reach of Bull River under the post redevelopment 2 m^3/s minimum summer flow release.	Ho1 is accepted. Ho2 is rejected Ho3 is accepted	Ho1:30-31 Ho2:30-31 Ho3: 25-29, 31-36, 39-43
		Is there an alternate minimum instream flow discharge that, in combination with the productive capacity realized from the	Ho_4: There is an alternate minimum instream flow that, in combination with the productive capacity realized from the compensation habitat, achieves the Aberfeldie Redevelopment	Ho4 is rejected if the alternate flow is <2 $m^3 \cdot s^{-1}$.	41-43

Project number	Objectives	Management questions	Management hypotheses	Status following work in 2012	Page(s) showing the result for ABFMON5-2a
		compensation habitat, achieves the Aberfeldie Redevelopment project compensation goal of no-net-loss of productive capacity in the diversion reach of the Bull River?	project compensation goal of no-net-loss of productive capacity in the diversion reach of the Bull River?		

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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) (Figure 1). 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 late March to late December annually but spill now occurs from late April to late July. During the new non-spill periods in March through April and August through December, all flow in the river can be diverted through the penstock, except for a minimum flow release to the diversion reach that is between the dam and powerhouse. Minimum flows were negotiated as part of water use planning (BC Hydro 2006). They are 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.

The diversion reach consists of a canyon segment and test segment (Figure 2). The 840 m long upstream "canyon segment" has moderate to high gradient, narrow steep walls, and substrate consisting of bedrock and large boulder. There are several barriers in the canyon that are impassable to fishes, 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". 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 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.

[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).

Perrin and Bennett (2013a) showed that in late summer, 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 including natural variation not related to maintenance outages at the powerhouse that can cause flow to pass through the diversion reach for the duration of maintenance. The combined density of mayflies, stoneflies, and caddisflies (commonly known as the EPT, that is an acronym for Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)) considered most sensitive to environmental change among aquatic invertebrates increased by 50% due to the change in flow in the diversion reach in late summer when biological production is considered relatively high compared to other times of the year. The response by EPT was hypothesized to be caused by lower hydraulic stress compared to conditions before the upgrade. The EPT density in the reach downstream of the powerhouse (called the downstream reach) did not vary with operations. 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.

To compensate for potentially lost habitat in the diversion reach at lower flows during new non-spill periods, 5,290 m^2 of side channel habitat was constructed in an area approximately 500 m downstream of the Aberfeldie generating station in 2009 (McPherson et al. 2010; Figures 1, 3, and 4). The area of the side channel was 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 upgrade to minimum flow afterwards. The side channel was constructed with substrate particle sizes to support benthic assemblages and provide complex spawning and rearing habitat for fishes. This productive capacity of the side channel is expected to at least equal or exceed amounts that were present in the diversion reach before the upgrade. This hypothesis was tested in this study by comparing invertebrate abundance in the test segment of the diversion reach before the upgrade to the sum of invertebrate abundance in the test segment and side channel after the upgrade.

The Water Use Planning Consultative Committee and Fisheries Technical Committee (BC Hydro 2006) 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 3 and 4 while 1 and 2 are answered in a companion report (Perrin and Bennett, 2013a). In this study, “productive capacity” is measured in terms of primary (algae) and secondary (invertebrates) productivity 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. Biomass is not included because it was not measured in the years before the Aberfeldie upgrade. 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 is consistent with the accepted definition of productive capacity for fishes 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).

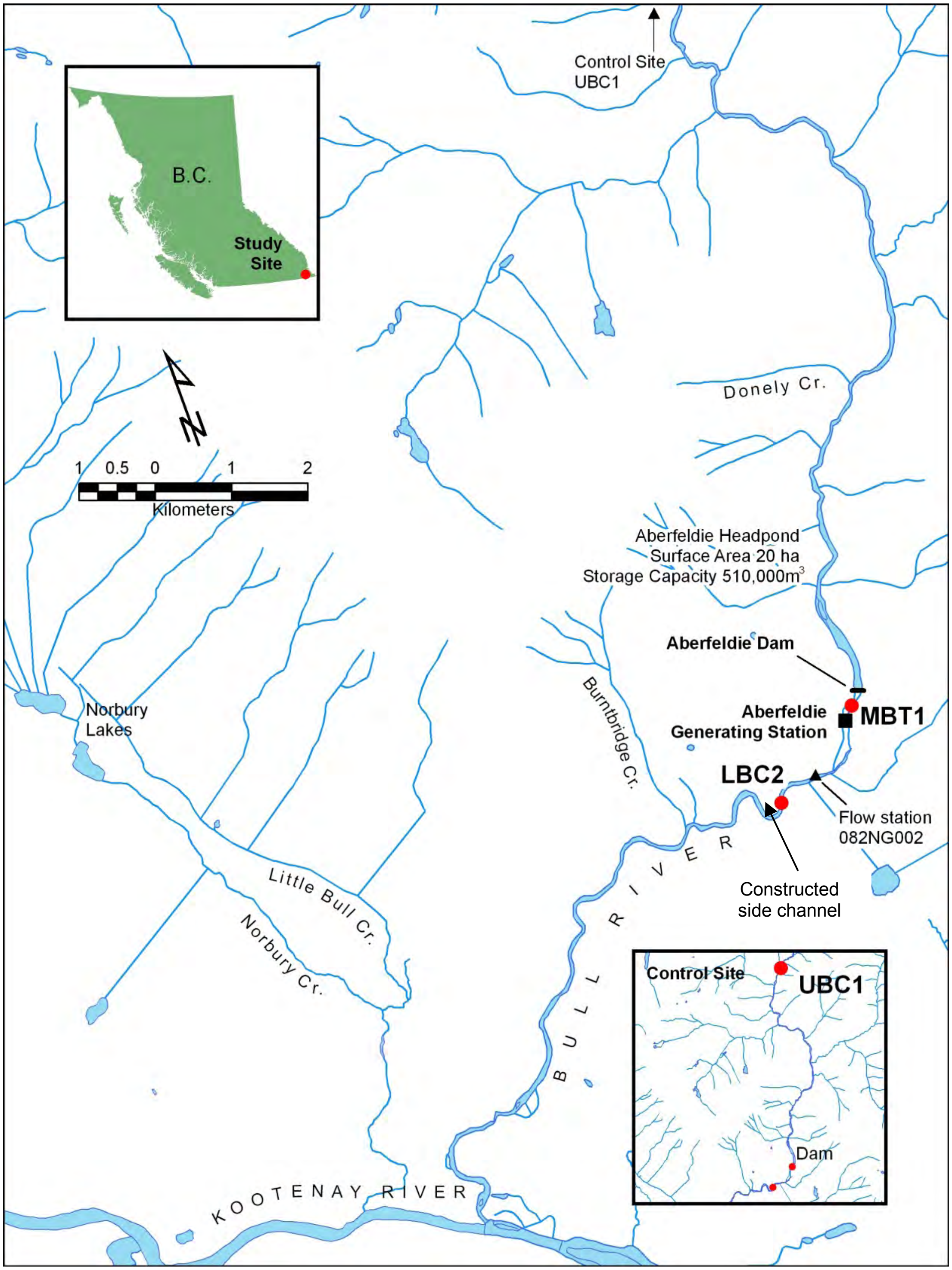


Figure 1. Bull River study area showing the geographic location and layout of the Aberfeldie generating facilities and sampling sites. UBC1 = Upper Bull Control site, MBT1 = Middle Bull Treatment site, LBC2 = Lower Bull Control site. The general area of the constructed side channel is marked with an arrow.

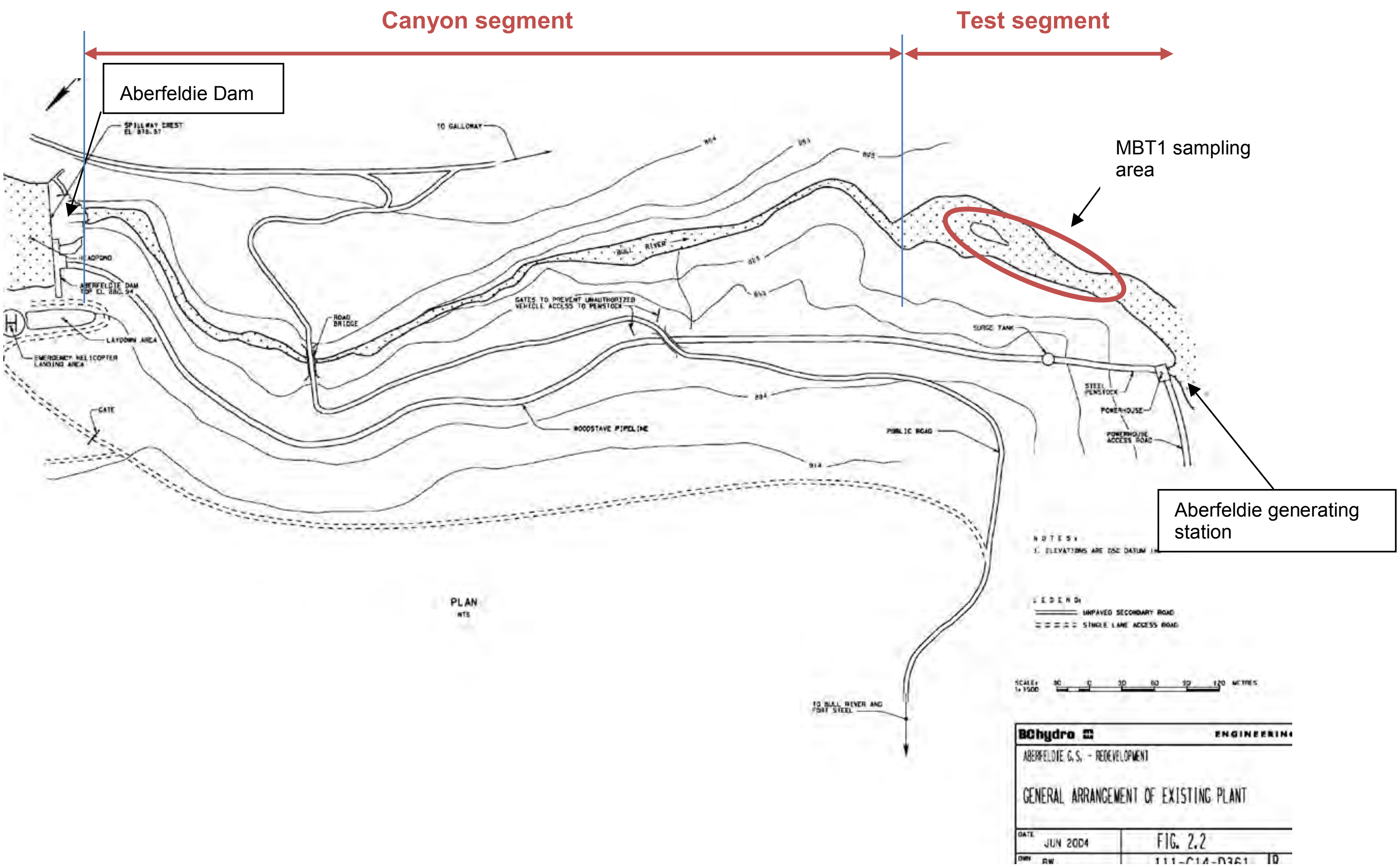


Figure 2. Map of diversion reach of the Bull River showing the canyon and test segments and the MBT1 sampling site. Map modified from BC Hydro (2004).

Figure 3. Image of the constructed side channel in 2009 looking upstream from the flow monitoring station. Flow in the channel at this time was 0.013 $m^3 \cdot s^{-1}$.

2 METHODS

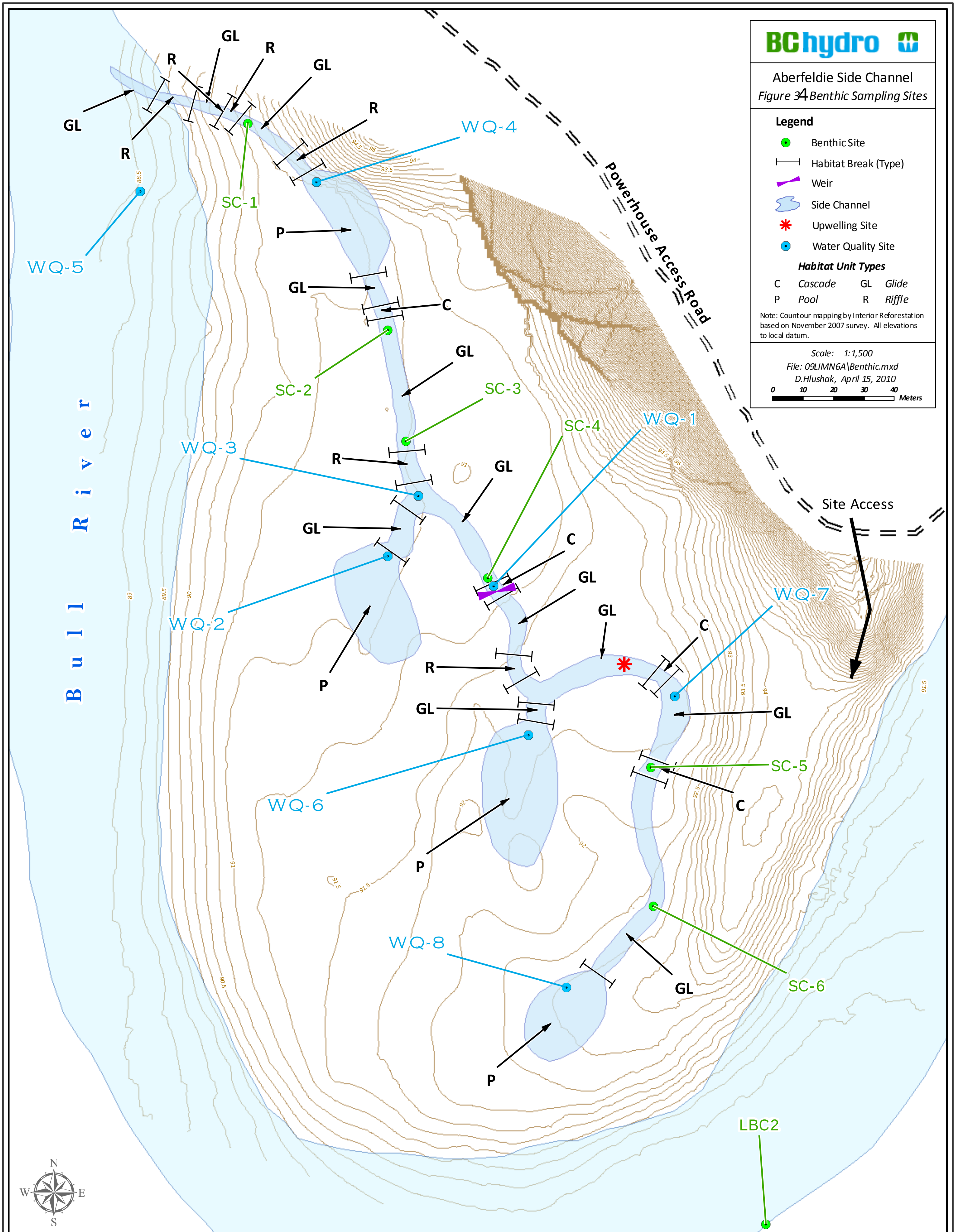
2.1 Study site

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 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.

Three sites were selected for sampling of water, periphyton and benthic invertebrates in the Bull River mainstem (Figure 1). An upstream control site was located 13.6 km upstream of the dam (UBC1), a treatment site was located in the test segment of the diversion reach (MBT1), and a third site was located downstream of the powerhouse (LBC2). They are described in detail by Perrin and Bennett (2013a).

The Aberfeldie side channel was constructed in September 2009 on the east floodplain of the Bull River across the river from the LBC2 sampling site (Figures 1, 3, and 4). The channel follows a low gradient of contours along the river floodplain where the river makes a 90º bend to the north. Groundwater supplies all flow to the channel. The channel is 585.5 m long and with wetted widths ranging from 1.3 m to 25.4 m, it provides 5,290.4 m^2 of aquatic habitat of which 2,164 m^2 is riffle (McPherson et al. 2010). The channel was designed to support various life stages of Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*), bull trout (*Salvelinus confluentus*), mountain whitefish (*Prosopium williamsoni*), and Kokanee (*Oncorhynchus nerka*). Gravel and other sediments were placed to support fish rearing and spawning and the production of periphyton and benthic invertebrates that may supply food for fish. The channel consists of glides, riffles, and pools with abundant cross channel woody debris but unstable banks. Design flows were 0.05 to 0.25 $m^3 \cdot s^{-1}$ but measured flows during monitoring in 2009 ranged from 0.004 to 0.036 $m^3 \cdot s^{-1}$ (McPherson et al. 2010). With these low flows, there is little hydrologic variation. Most habitats have standing water (Figure 5) connected by small riffles (Figure 6). Other details regarding channel habitat and initial use by fishes are reported by McPherson et al. (2010).

Six side channel sites labelled SC1 at the downstream end through SC6 at the upstream end were sampled in 2009, 2010 and 2012 (Figure 4). They were distributed among the two primary flow types; SC3, SC4, and SC5 were in active flowing water that was limited to short 1-4 m segments between more extensive standing water where sites SC1, SC2, and SC6 were located. All sites were easily accessed via a foot trail from the adjacent powerhouse access road.



BC hydro
Aberfeldie Side Channel
Figure 34 Benthic Sampling Sites
Legend
Benthic Site
Habitat Break (Type)
Weir
Side Channel
Upwelling Site
Water Quality Site
Habitat Unit Types
C Cascade GL Glide
P Pool R Riffle
Note: Countour mapping by Interior Reforestation based on November 2007 survey. All elevations to local datum.
Scale: 1:1,500
File: 09LIMN6A\Benthic.mxd
D.Hlushak, April 15, 2010
0 10 20 30 40 Meters
Powerhouse Access Road
Site Access
Bull River
WQ-1
WQ-2
WQ-3
WQ-4
WQ-5
WQ-6
WQ-7
WQ-8
SC-1
SC-2
SC-3
SC-4
SC-5
SC-6
LBC2

Figure 5. Side channel sampling site number SC2 showing near-standing water. The white object in the centre of the channel is a periphyton sampling plate that is described in Section 2.3.

Figure 6. Side channel sampling site number SC3 showing flowing water.

2.2 Wetted area in the test segment of the diversion reach before and after the upgrade

A comparison of area survey data using modeling (Cope 2005) and direct measurement (Perrin and Canning 2010) showed large differences in wetted areas for equivalent flows (Perrin and Canning 2010). Among comparisons of wetted area at 0.25 $m^3 \cdot s^{-1}$, 0.5 $m^3 \cdot s^{-1}$, and 2 $m^3 \cdot s^{-1}$, the measured values exceeded the modeled estimates by 281 – 424%. Given that the modeled estimates were extrapolated beyond the range of calibration measurements that included only two ground transects, the direct measurements were considered most reliable (Perrin and Canning 2010). Even the direct measurements in the field that were used for calibrating the model by Cope (2005) are suspect because they were based only on the two ground transects compared to the nine used in the same riffle area in 2009. An example of the error is shown in Table 1 in which the wetted area measured using nine transects at 2 $m^3 \cdot s^{-1}$ in 2009 was greater than area based on the two transects at 6.4 $m^3 \cdot s^{-1}$ from 2004.

Given the uncertainty associated with modeled areas at flows before the upgrade, non-linear regression was used to fit a curve to the three observations of flow and measured wetted area from 2009 (Figure 7). The resulting equation 1 was used to calculate area wetted by the pre-upgrade median flow of 6 $m^3 \cdot s^{-1}$ (Perrin and Bennett 2013a). There are two sources of error with this approach; a high fit of the curve to the data due to a very low number of data points (n=3), and error associated with extrapolation of values outside of the range of values used to fit the curve. However, this option was the best available for determining pre-upgrade wetted area at flows greater than 2 $m^3 \cdot s^{-1}$ given the absence of reliable measured areas at pre-upgrade flows.

Table 1 Measured wetted riffle area in the test segment of the diversion reach at various flows.

Survey date	Mean daily flow release to the diversion reach ($m^3 \cdot s^{-1}$)	Surveyed wetted riffle area in test segment of the diversion reach (m^2)	Data source
4-Nov-2004	11.0	2343	Cope 2005.Table 3.5
4-Nov-2004	6.4	1979	Cope 2005.Table 3.5
30-Nov-2004	4.4	1786	Cope 2005.Table 3.5
30-Sep-2009	2.0	2025	Perrin and Canning 2010
1-Oct-2009	0.5	1484	Perrin and Canning 2010
2-Oct-2009	0.25	1180	Perrin and Canning 2010

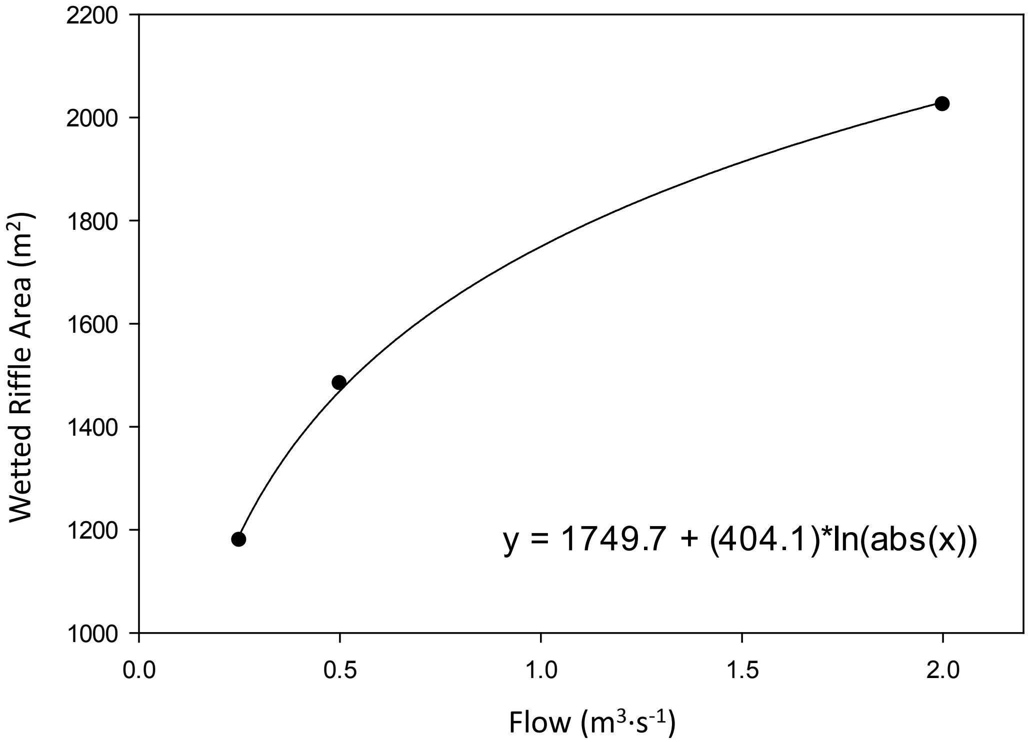


Figure 7 Wetted riffle area as a function of flow in the test segment of the diversion reach using measurements from 2009 (Perrin and Canning 2010).

$$y = 1749.7 + (404.1) * ln\big(abs(x)\big) \quad \text{Equation 1}$$

Where:
y = wetted riffle area (m^2), and
X= flow ($m^3{\cdot}s^{-1}$)

2.3 Biological field and laboratory procedures

Artificial substrata called “periphyton plates” were used to sample periphyton assemblages (Figure 5). 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 has a rough texture that allows for rapid seeding by algal cells and the adhered biomass is easily sampled (Perrin et al. 1987).

Periphyton biomass was sampled weekly from each of the six side channel sites over an incubation period of seven weeks beginning in the first or second week of August in each of 2009, 2010 and 2012. Sampling was synchronized each year with periphyton sampling at the three Bull River sites that is described by Perrin and Bennett

(2013a). The plates were submerged in both the flowing and standing water segments of the side channel. Each sample consisted of a 2 cm diameter core of the Styrofoam and the adhered biomass removed 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 chl-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). The samples collected over 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.

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 collection net and removable cod end. 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 approximately 1 m upstream to another random location and the sample collection was repeated. Contents accumulated in the sampler net after five placements were sampled constituted a single sample. Total surface area for a single sample was 4500 cm^2 (900 cm^2 sampler area x 5 composited placements of the sampler). One sample was collected from each of the six side channel sites. The samples were preserved in 10% formalin immediately after collection.

In the laboratory, each invertebrate 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 genus or lowest reliable taxonomic level 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 Habitat measurements

Measurements of habitat attributes were made to support analyses to show most important attributes that may explain differences of biological assemblages between the river and side channel. The method of measuring these attributes is described as follows.

Daily mean flow in the Bull River at LBC2 was obtained from the Water Survey of Canada (WSC) for station number 08NG002 located between the generating station and LBC2 (Figure 1). Mean daily flows at UBC1 were determined as:

$$Q_i = Q_r(\frac{W_i}{W_r}) \qquad \text{Equation 2}$$

where Q is daily mean flow at site i (UBC1 in this case) or the reference site r (WSC station 08NG002) and W is watershed area determined from data in the Province of BC, Integrated Land Management Bureau, Land and Resource Data Warehouse (LRDW). Watershed Atlas polygons were manually altered for UBC1 based on interpretation of provincial TRIM contours. Mean daily flow at MBT1 was accessed from BC Hydro Power Records as water released from the Aberfeldie dam.

Flow in the side channel was determined by recording of stage height off a staff gauge that was installed on a rectangular weir (Figure 3) every five to thirty days. Fifty-five gauge readings were recorded in 2009, 2010 and 2012. Flow was determined from a rating curve developed for the weir. Flow between days of stage records was linearly interpolated.

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 Bull River site and at sites SC2 and SC5 in the side channel. Daily mean temperature at the other side channel sites was determined by linear interpolation between upstream and downstream ends of the side channel. 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 of the three Bull River sites and from side channel sites SC1 (downstream end) and SC6 (upstream end) at the start and finish of the periphyton sampling time series 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. Caution was used when interpreting the presence of ammonium in these blanks. The blanks are acidic and they tend to scavenge ammonia from the air, particularly during filtration. This ammonia scavenging may not occur in a normal water sampled that has a pH that is more circumneutral or slightly alkaline. Hence, ammonium in blanks was not considered evidence of sample 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):

$$D_f = \left(\frac{A-B}{(A+B)/2}\right) * 100 \qquad \text{Equation 3}$$

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. Resolution and accuracy data as follows are reported on the YSI website at (http://www.ysi.com/productsdetail.php?6920-V2-3). The dissolved oxygen sensor had a resolution of 0.01 $mg \cdot L^{-1}$ and accuracy of ±0.1 $mg \cdot L^{-1}$. The turbidity sensor had a resolution of 0.1 NTU and accuracy of ±0.3 NTU. The total dissolved solids was calculated internally from a conductivity sensor that had a resolution of 0.001 – 0.1 $mS \cdot cm^{-1}$ (range dependent) and accuracy of 0.001 $mS \cdot cm^{-1}$. The pH sensor had a resolution of 0.01 units and accuracy of ±0.2 units.

Descriptions of other measurements including water depth and velocity and coding for various habitat conditions are listed in Table 2.

Table 2. List of habitat attributes and methods of measurement in the Bull River and the constructed side channel.

Habitat attribute	Method of measurement
Position and elevation of each site	Recorded on a Garmin GPSmap 76CSx receiver. The instrument has measurement accuracy of ±3 m.
Current velocity	Measured weekly at the upstream edge of each periphyton plate using a Swoffer Instruments (Seattle WA) velocity sensor
Water depth	Measured weekly as the interval between at the top surface of each periphyton plate and the water surface using a standard meter stick
Percent composition of pool, glide, riffle, and cascade	Visually estimated at main stem sites over 100 m of river length at the sampling site. Habitat composition for the side channel was calculated as percent composition by habitat type over a distance 50 m upstream of a sampling site as reported by McPherson et al. (2010).
Dominant habitat type	Habitat type (pool, glide, riffle, or cascade) occurring in the highest percentage over 50 m upstream of a given sampling site.
Channel type	Coded 0 for constructed channel and 1 for the Bull River.
Wetted and bankfull widths	Measured with an Opti-Logic model 600XT range finder at three transects of each river site. Average values were assigned as the wetted and bankfull width for each site. Wetted and bankfull widths in the side channel were those reported by McPherson et al. (2010).
Embeddedness	Visually estimated using a score of 1 to 5 where 1 = not embedded, 2 = 25% embedded, 3 = 50% embedded, 4 = 75% embedded, 5 = completely embedded.
Channel cover type	Woody debris, boulder, undercut bank, deep pools, and overhanging vegetation were coded as 1 = none, 2 = trace, 3 = moderate, 4 = abundant based on visual observation
Dominant riparian class	Coded as 1 = unvegetated, 2 = grass or herb, 3 = shrub, 4 = deciduous forest, 5 = coniferous forest, 6 = mixed forest
Structural stage of riparian vegetation	Coded as 1 = less than 5% cover, 2 = shrub or herb with <10% cover, 3 = pole sapling stage 15-20 years old, 4 = young forest 30-80 years old, 5 = mature forest with well developed understory.
Riparian canopy closure over the stream	Coded as 1 = 0%, 2 = 1-25%, 3 = 26-50%, 4 = 51-75%, 5 = 76-100%.
Local erosion	Coded as 1 = eroding banks apparent at sampling site or 0 = no eroding banks present at sampling site.

Habitat attribute	Method of measurement
Upstream erosion	Coded as 1 = episodic turbidity events observed at the sampling site, indicating upstream slope failure or erosion event, 0 = no episodic turbidity event observed or reported.
Proximity to a dam or stream source	Coded as 1 = within 1 km and no major tributary between the dam or origin and sampling site, 2 = 2-5 km, 3 = 5-10 km, 4 = no dam upstream or > 10 km from origin or a large tributary enters upstream of the sampling site. This coding was used as a surrogate for invertebrate recruitment to the sampling site.

2.5 Data analysis

2.5.1 Quality assurance

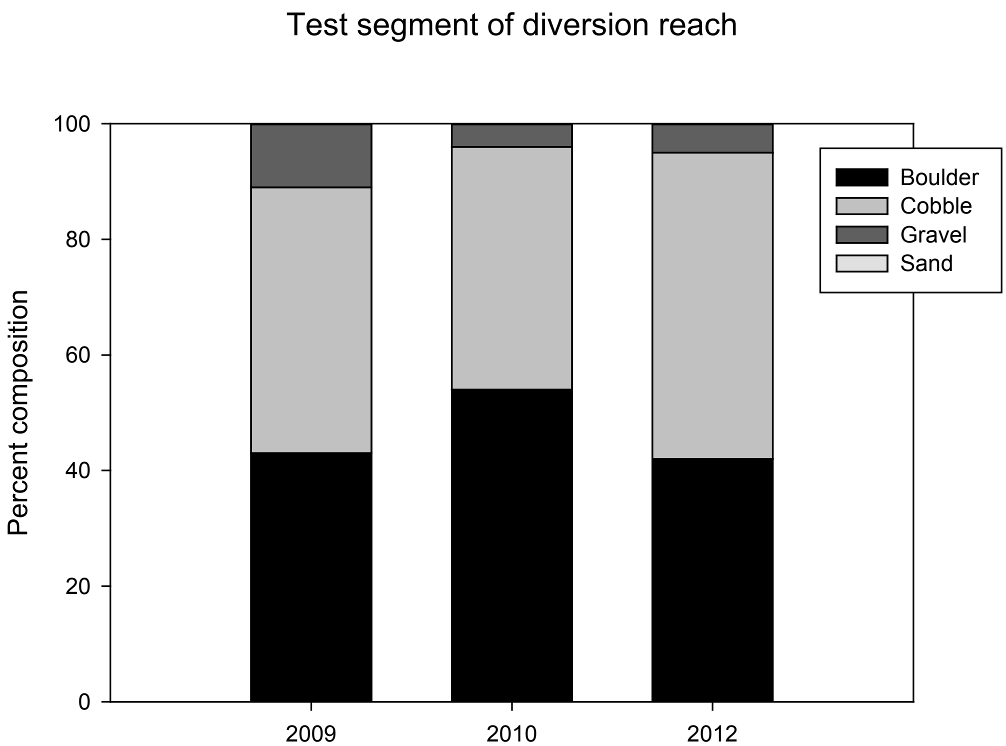
Following original data entry by the second author the senior author performed quality checks. Every tenth row of each data sheet that appears as a digital appendix to this report was checked against raw field data sheets or lab reports. If errors were found, the second author checked complete sections 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 Wetted usable area

Total wetted area of non-pool habitat in the test segment of the diversion reach was determined using methods described in Section 2.2. That total area included all particle sizes of substrate. However, boulders and bedrock that were part of the substrate in the test segment may not be favoured by invertebrates as compared to the smaller cobbles and gravels that were also common. To provide an estimate of minimum abundance of invertebrates in the test segment of the diversion reach, the Wolman pebble count data were compiled for 2009, 2010 and 2012 to estimate the proportion of particles smaller than boulder (<256 mm, Wentworth 1922) that were considered usable by benthos in the test segment. The total wetted area was multiplied by the proportion of substrates <256mm to provide an estimate of wetted usable habitat area. An assumption in the calculation was that each measured particle, regardless of size, occupied an equal area of wetted habitat, which may not be true so wetted usable area among particles <256 mm was overestimated. Another assumption was that invertebrates did not use boulders and bedrock. While densities are expected to be lower on the boulders compared to smaller substrates, invertebrates are not likely to be completely absent. Finally, the method assumes there was no change in particle size distribution under the two flow scenarios, which was verified during the site visits (Figure 8).

Figure 8. Particle size distribution in the test segment of the diversion reach determined from Wolman 100 pebble counts.



2.5.3 Biological assemblages in the Bull River and side channel

The focus of analyses was on benthic invertebrates because they provide 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 cutthroat trout (Schoby and Keeley 2011) and whitefish (McPhail and Troffe 1998) that are present in the Bull River (Cope 2005).

Multivariate tests were run to examine spatial and temporal variation among assemblages between the Bull River sites and the side channel. This analysis was for years after the Aberfeldie upgrade when the side channel was operating. The side channel was not operating before the upgrade. Invertebrate counts were compiled with site (UBC1, MBT1, LBC2 and SC) and year (2010 and 2012) coded as a factors to facilitate several types of analysis. Data from 2009 were not used because the minimum flow conditions were not met in the diversion reach in 2009 (Perrin and Bennett 2013a). 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. Some taxa were clumped: their abundance 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 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 increasing severity 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 24 (2 years x 3 sites x 4 samples) to 6 (2 years x 3 sites x 1 average value) in the diversion reach and from 12 (2 years x 6 samples) to 2 (2 years x 1 average value) in the side channel.

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 among assemblages between Bull River locations and the side channel (4 sites x 2 years = 8 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 spatial patterns in the count data.

Site effects were tested by one way analysis of similarity (ANOSIM) on the same dispersion weighted invertebrate count data used in the NMDS. An overall site effect was interpreted from the global ANOSIM R statistic that varies from 0 (no site effect on assemblages) to 1 (dissimilarities of assemblages between sites were greater than dissimilarities within sites). Significance was tested by standard permutation in Primer (Clarke and Gorley 2006). If the global R statistic was greater than 0.3 and significant, pairwise R statistics were interpreted to determine which pairs of sites were most different from one another. 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 within a site and >90% of dissimilarities of assemblages between sites.

Periphyton assemblages were compared between stations and times using non-metric multidimensional scaling on dispersion weighted algal cell biovolume data in PRIMER. The ordination showed dissimilarities of periphyton assemblages between samples with coding by site and year. Several measures of periphyton cell biovolume and diversity were summarized by site for the period after the upgrade (2010 and 2012). They included peak biomass, species richness (number of unique species per sample) and Simpson's Index of Diversity (Krebs 1999). A site effect on algal peak biomass (PB) was tested using a one-way analysis of variance (ANOVA). The ANOVA was followed by Tukey's test to examine post hoc comparisons between sites for the metric. Significance was set at p=0.1. All ANOVAs and the Tukey tests were run in Systat v11 (Systat 2004).

2.5.4 Invertebrate abundance in the diversion reach and side channel

Two processes occurred to modify benthic assemblages in the diversion reach among years before and after the Aberfeldie upgrade. One is change in density within continuously wetted areas and the other is change in wetted area hosting benthic invertebrates. Invertebrate abundance within the whole test segment of the diversion reach was the product of wetted area and density before and after the upgrade. Calculation of wetted area was explained in Section 2.2. This calculation was run for invertebrates from all taxa, chironomids, and the EPT (Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)). There was no effect of the Aberfeldie upgrade on density of all invertebrates and chironomids within the benthos assemblage and a significant 50% increase in the density of the EPT (Perrin and Bennett 2013a). Further statistical tests of the effect of the upgrade on benthos metrics were not necessary.

Given no significant difference in the density of all invertebrates and the chironomids due to the upgrade, all four years (two years before and two years after) of density data were combined to determine the mean density of all invertebrates and chironomids in both the before and after periods. Only wetted area affected abundance of all invertebrates and chironomids.

For the EPT, abundance before the upgrade was the product of wetted area and density in the test segment of the diversion reach before the upgrade and abundance after the upgrade was the product of wetted area and density after the upgrade. Separate densities before and after the upgrade were used because EPT density was found to significantly increase due to the upgrade (Perrin and Bennett 2013a).

Arithmetic mean abundance and standard deviation were calculated for the “before” and “after” periods for each metric. The average density for each metric from the four samples from each site in the Bull River or the six samples collected from the side channel in each year was considered a single observation to avoid pseudoreplication. That average density was multiplied by wetted area to yield the abundance values. The mean and standard deviation was calculated after the product of wetted area and density was determined, which means that each observation was abundance in the whole test segment of the diversion reach or in the whole side channel. For all invertebrates and the chironomids, year was considered a replicate (n=4; 2005, 2006, 2010, 2012). For the EPT, n=2 (one observation from 2005 and one from 2006 for the before period and one observation from 2010 and one from 2012 for the after period). For the side channel n=2 (2010 and 2012).

2.5.5 Biotic – abiotic matching

Attributes of habitat that may be important in influencing biological assemblages in the main stem and side channel were found using the procedure called BEST (Clarke and Gorley 2006). The starting list of habitat variables included all those described in Section 2.4. Variables considered redundant with more relevant variables were deleted. A resemblance matrix of Bray Curtis dispersion weighted invertebrate counts was compared with a corresponding resemblance matrix of Euclidean distance environmental data. Before running the distance calculations, skewness in distributions between environmental variables was corrected using the log(x+1) transformation and environmental data were normalized (subtract the mean and divide by the standard deviation for each variable) to correct for different scales of measurement or coding. Matching between the two matrices was determined with the Spearman coefficient (ρ) and the best combination of six habitat variables that optimized ρ in the comparisons of biological and habitat variables was determined in Primer v6.

3 RESULTS

3.1 Wetted usable area

Mean daily flow in the side channel was lowest in 2012 (0.008 $m^3 \cdot s^{-1}$) and greatest in 2010 (0.014 $m^3 \cdot s^{-1}$), although there was little variation between and within years (Table 3). The coefficient of variation of flow during the sampling periods was <0.4

in all three years. Flows in the channel were underestimated in 2012 due to a leak in the weir noted at the beginning of the sampling period. Flows in the side channel were <0.3% of flows in the diversion reach during comparable years (Tables 3 and 4).

Table 3. Flow in the Aberfeldie side channel during the sampling period (August and September) in 2009, 2010 and 2012.

Year	Mean daily flow in the side channel ± SD ($m^3 \cdot s^{-1}$)	Median flow ($m^3 \cdot s^{-1}$)	Coefficient of variation	Range of daily flows in the side channel ($m^3 \cdot s^{-1}$)
2009	0.011 ± 0.004	0.012	0.38	0.006 – 0.016
2010	0.014 ± 0.003	0.014	0.22	0.010 – 0.020
2012	0.008 ± 0.003	0.006	0.33	0.006 – 0.012

Table 4. Water releases to the diversion reach during the sampling period (August and September) before and after the Aberfeldie upgrade (data from Perrin and Bennett 2013a).

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 ($m^3 \cdot s^{-1}$)
Before (2005 and 2006)	7.5 ± 4.6	6.3	0.62	2.4 – 23.5
After (2009, 2010, 2012)	7.5 ± 12.5	2.0	1.67	0.5 – 44.8
After (excluding 2009 when an extended maintenance outage occurred)*	3.3 ± 5.7	2.0	1.75	0.5 – 36.0

*see Perrin and Bennett (2013a) for details.

The median daily flow released to the diversion reach was 6.3 $m^3 \cdot s^{-1}$ in the study periods during the years before the upgrade (Perrin and Bennett 2013a). Using Equation 1, flows of 6.3 $m^3 \cdot s^{-1}$ wetted 2,494 m^2 of non-pool habitat. After the upgrade, the median daily flow release to the diversion reach was equal to the prescribed minimum daily flow of 2 $m^3 \cdot s^{-1}$, which wetted 2,025 m^2 of non-pool habitat (Perrin and Canning 2010). Therefore, a flow reduction from 6.3 $m^3 \cdot s^{-1}$ to 2 $m^3 \cdot s^{-1}$ resulted in a loss of 469 m^2 or 19% of wetted non-pool habitat area in the diversion reach. An area of

1,564 m^2 of non-pool habitat was constructed in the side channel (McPherson et al. 2010). The total area of non-pool habitat in the diversion reach and side channel was 4,189 m^2 at minimum flow after the upgrade (Table 5). The additional habitat provided in the side channel (1,564 m^2 in wetted riffle area) resulted in an overall gain in habitat of 1,095 m^2, a 44% increase from the pre-upgrade wetted area.

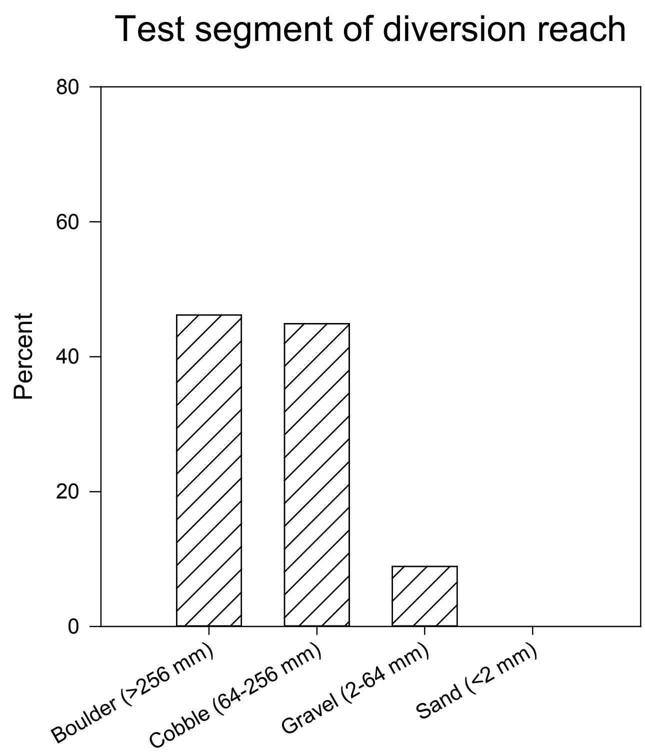
Table 5. Flow and wetted non-pool habitat area in the diversion reach and side channel before and after the Aberfeldie upgrade.

Period	**Median daily flow release to the diversion reach ($m^3 \cdot s^{-1}$)**	**Wetted non-pool area in the test segment of the diversion reach (m^2)**	**Wetted non-pool area in the side channel (m^2)**	**Total wetted habitat area (sum of test segment and side channel) (m^2)**
Before	6.3	2,494[a]	Not applicable	2,494
After	2.0	2,025[b]	1,564	3,589

[a] Non-pool area (all riffle in the test segment of the diversion reach) estimated using regression (equation 1) developed with data from three surveys in 2009 by Perrin and Canning (2010).

[b] Direct measurement at 2 $m^3 \cdot s^{-1}$ (from Perrin and Canning 2010)

Boulder and bedrock accounted for 46% of substrates in the test segment of the diversion reach and 5% of substrates in non-pool habitat of the side channel (Figure 9). Wetted usable habitat area was the product of the proportion of substrates smaller than boulders (i.e. <256 mm) and wetted non-pool area. This calculation resulted in 1,347 m^2 of usable non-pool habitat in the test segment of the diversion reach before the upgrade (median flow of 6.3 $m^3 \cdot s^{-1}$), 1,094 m^2 of usable non-pool habitat in the test segment of the diversion reach after the upgrade (median flow of 2 $m^3 \cdot s^{-1}$), and 1,486 m^2 of usable non-pool habitat in the side channel.



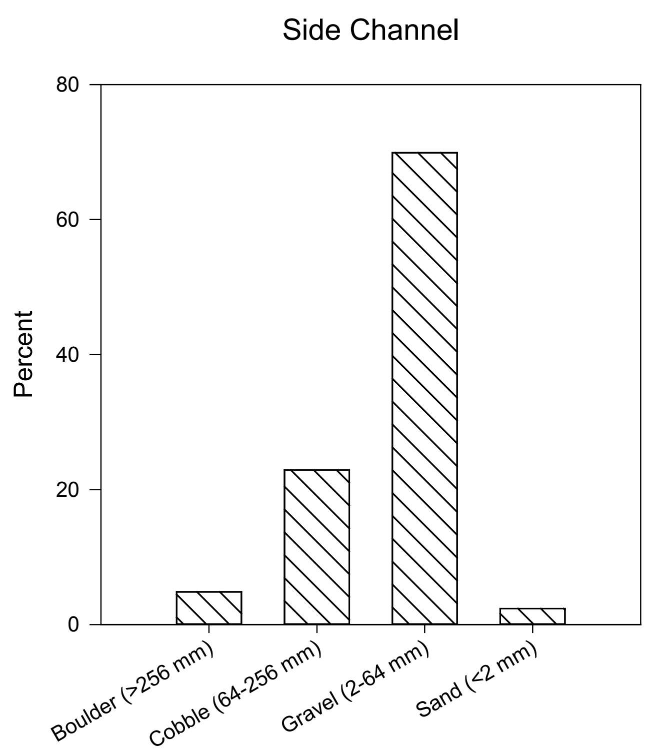


Figure 9. Particle size distribution in sediment of the test segment of the diversion reach (top) and the constructed side channel (bottom).

3.2 Biological assemblages in the Bull River and side channel

3.2.1 Invertebrates

A frequency distribution showed many invertebrate taxa had high D (Dispersion) values (many well over 10 and some approaching 2,000) indicating extreme clumping. This result justified application of dispersion weighting to downweight taxa having the greatest degree of clumping prior to running multivariate analyses.

Assemblages in the side channel were different from those of the main stem (Figures 10 and 11) (ANOSIM Global R = 0.42, p = 0.095). R values from the pairwise tests showed complete dissimilarity of assemblages between the side channel and Bull River sites (Table 6), which confirmed the ordination (Figure 10). Taxa contributing to 90% of the dissimilarities were the EPT, chironomids, non-chironomid dipterans (mainly Simulium sp.), naidid worms, and ostracods, with minor contributions from other taxa (Tables 7 and 8). Over half the dissimilarities were due to differences in densities of chironomids, Baetis, Capniidae and Simulium. The side channel lacked EPA taxa that were in the Bull River, particularly Baetis sp. and Capniidae. Twelve of the EPT taxa that were present in the Bull River were not found in the side channel. In contrast the side channel had an abundance of Simulium sp. that was almost absent from the test segment and was absent from UBC1. Ostracods were not present in the Bull River but were present in the side channel. These main differences would have contributed most to the highly clumped distribution of samples between the side channel and Bull River on the ordination in Figure 10.

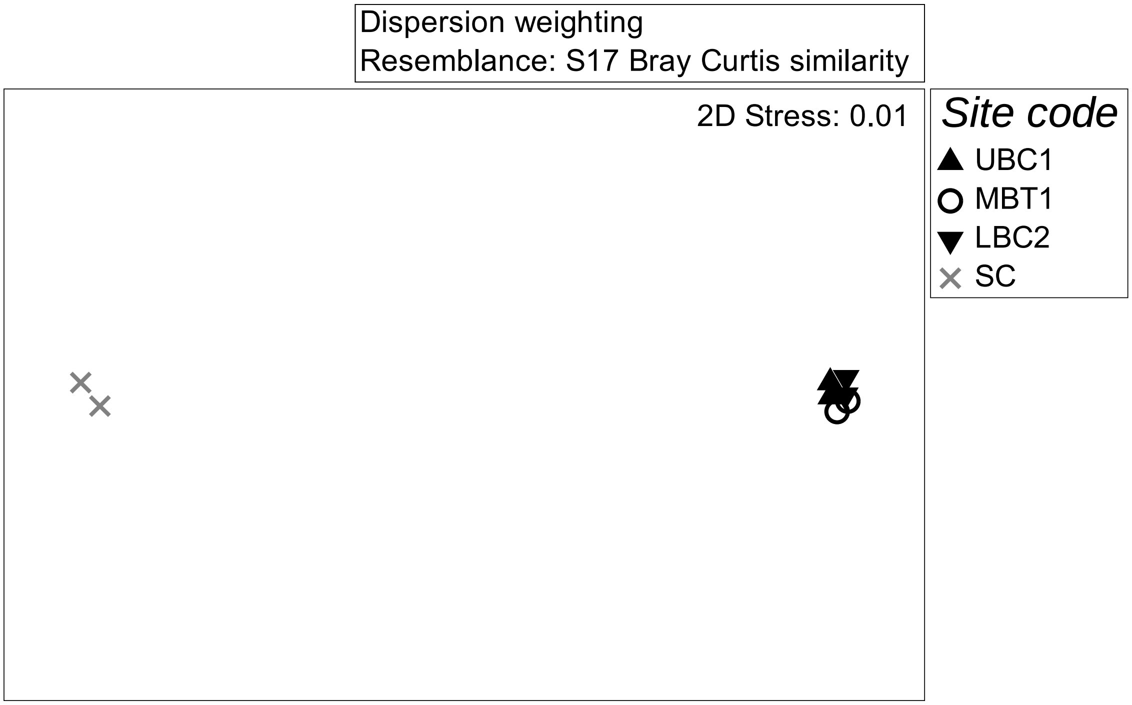


Figure 10. Non-metric multidimensional scaling ordination of genus-level benthic invertebrate samples from the Bull River and the side channel (SC) after the Aberfeldie upgrade (2010 and 2012).

Table 6. Pair-wise ANOSIM R statistics for contrasts between pairs of sites on the Bull River and side channel (SC) after the Aberfeldie upgrade (2010 and 2012). ANOSIM was run on benthic invertebrate count data averaged by site and year. The global R for site was 0.42 (p=0.095).

Factor	**Pairs tested in ANOSIM**	**R statistic**	**p value**
Location	UBC1, MBT1	0	0.67
	UBC1, LBC2	0	0.67
	MBT1, LBC2	-0.25	0.67
	UBC1, SC	1	0.33
	MBT1, SC	1	0.33
	LBC2, SC	1	0.33

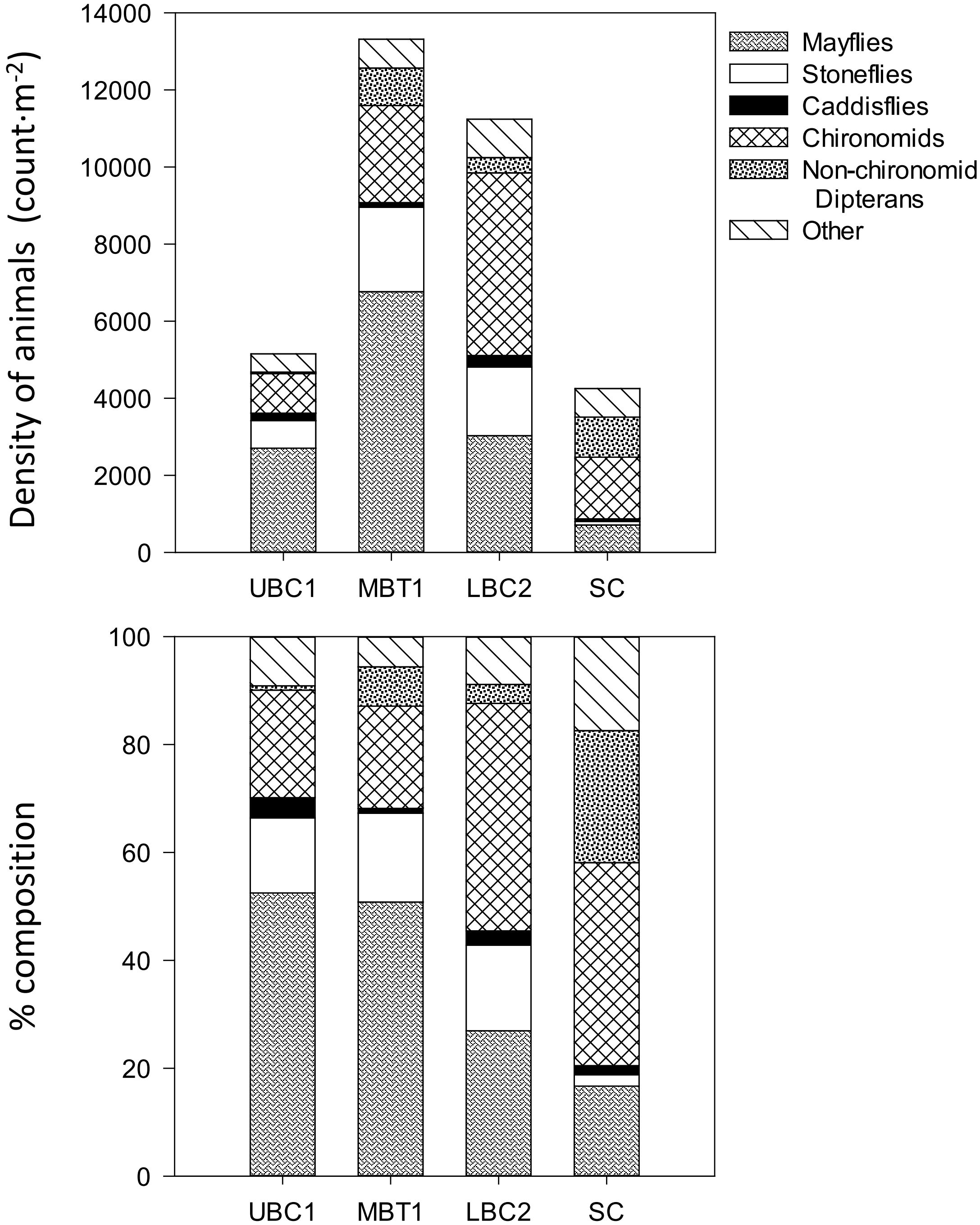


Figure 11 Density of benthic invertebrates by taxonomic group at Bull River and side channel (SC) stations averaged for the sampling periods after the Aberfeldie upgrade (2010 and 2012), excluding 2009 when an extended maintenance outage would have confounded the test of change in flow (Perrin and Bennett 2013a). The “other” group included Acari, Ostracoda, Nemata, Planariid worms, Oligochaetes, Megaloptera, Hemiptera, Collembola, Coleoptera and Hydra.

Table 7. Density of taxa shown by SIMPER to cumulatively explaining 90% of dissimilarities of assemblages between each Bull River site (UBC1, MBT1, LBC2) and the side channel

(SC) after the upgrade. The metric to which each taxon is a part is shown for reference. DNC means the taxon ‘did not contribute’ to 90% of dissimilarities between the Bull River site and SC.

Genus or other classification	**Metric to which the taxon belongs**	**Mean density at UBC1 (number·m^{-2})**	**Mean density at MBT1 (number·m^{-2})**	**Mean Density at LBC2 (number·m^{-2})**	**Mean density at SC (number·m^{-2})**
Average dissimilarity to SC		64.4%	73.7%	69.3%	
Baetis	EPT	4415	13489	2739	1241
Chironomidae	Chironomids	2241	5484	10336	3533
Capniidae	EPT	395	2496	2584	20
Simulium	Total	0	14	233	2170
Naididae	Total	343	613	340	837
Zapada	EPT	389	1401	488	177
Ephemerella sp.	EPT	165	563	1153	0
Rhithrogena sp.	EPT	655	232	182	0
Torrenticolidae	Total	295	437	879	1
Nemouridae	EPT	440	254	163	0
Antocha	Total	DNC	1500	409	2
Ostracoda	Total	0	0	0	273
Doddsia	EPT	DNC	438	280	0
Heptageniidae	EPT	DNC	179	759	0
Cinygmula	EPT	259	DNC	289	36
Ephemerellidae	EPT	DNC	DNC	736	0
Drunella sp.	EPT	170	DNC	181	0
Parapleptophlebia	EPT	0	0	DNC	193
Hygrobatidae	Total	0	DNC	0	171
Lebertia	Total	124	139	245	13
Trombidiformes	Total	66	199	132	18
Taeniopterygidae	EPT	106	DNC	134	0
Epeorus	EPT	225	DNC	DNC	0
Diphetor	EPT	DNC	DNC	570	5
Testudacarus	Total	DNC	DNC	247	0
Rhyacophila	EPT	DNC	DNC	227	0
Spercon	Total	DNC	DNC	287	91
Wiedemannia	Total	DNC	408	DNC	0
Arctopsyche	EPT	168	DNC	DNC	0
Brachycentrus	EPT	142	DNC	DNC	0
Polycelis	Total	1	DNC	DNC	106
Sweltsa	EPT	88	DNC	DNC	0

Table 8. SIMPER output showing percent contribution of taxa cumulatively explaining 90% of dissimilarities of assemblages between UBC1, MBT1, LBC2 and SC after the upgrade. The metric to which each taxon is a part is shown for reference.

Genus or other classification	**Metric to which the taxon belongs**	**Percent contribution to dissimilarity between UBC1 and SC**	**Percent contribution to dissimilarity between MBT1 and SC**	**Percent contribution to dissimilarity between LBC2 and SC**
Average dissimilarity to SC		64.4%	73.7%	69.3%
Baetis	EPT	24.7	34.1	8.3
Chironomidae	Chironomids	11.6	9.7	25.8
Capniidae	EPT	3.7	13.6	14.0
Simulium	Total	15.7	8.1	8.3
Naididae	Total	5.3	2.3	2.9
Zapada	EPT	2.2	4.8	2.1
Ephemerella sp.	EPT	1.1	1.7	4.2
Rhithrogena sp.	EPT	4.9	1.3	<1
Torrenticolidae	Total	1.9	1.1	3.3
Nemouridae	EPT	3.8	1.3	<1
Antocha	Total		3.9	1.4
Ostracoda	Total	2.4	1.2	1.3
Doddsia	EPT		2.5	1.6
Heptageniidae	EPT		<1	2.6
Cinygmula	EPT	1.7		<1
Ephemerrelidae	EPT			2.6
Drunella sp.	EPT	1.3		<1
Parapleptophlebia	EPT	1.3	<1	
Hygrobatidae	Total	1.3		<1
Lebertia	Total	<1	<1	1.1
Trombidiformes	Total	<1	1.1	<1
Taeniopterygidae	EPT	1.1		<1
Epeorus	EPT	1.6		
Diphetor	EPT			1.9
Testudacarus	Total			1.3
Rhyacophila	EPT			1.1
Spercon	Total			<1
Wiedemannia	Total			
Arctopsyche	EPT	1.2		
Brachycentrus	EPT	1.0		
Polycelis	Total	<1		
Sweltsa	EPT	<1		

3.2.2 Periphyton

The periphyton assemblage in the side channel was different each year, but followed a pattern over time similar to that of Bull River assemblages (NMDS, Figure 12). In each year, side channel assemblages were different from the Bull River assemblages, which were more similar to each other within years than between years.

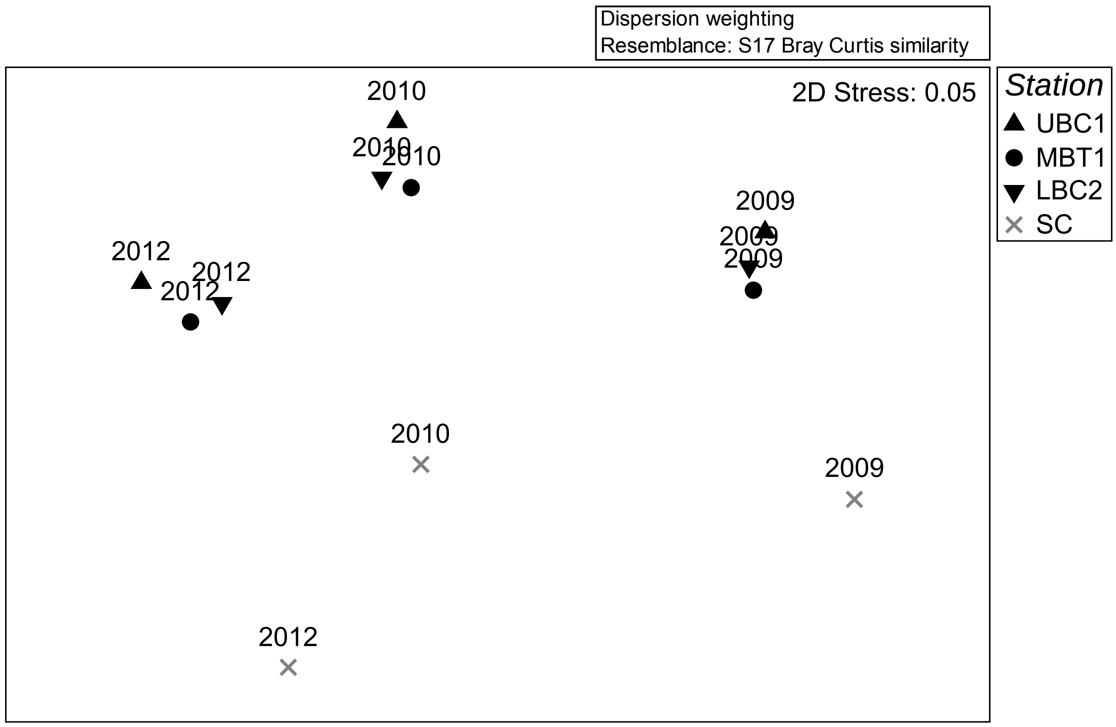


Figure 12 NMDS ordination of genus-level periphyton cell biovolumes averaged by site and year in the Bull River and side channel (SC).

Overall, the assemblage in the side channel was diverse and included more taxa from non-diatom divisions than in the Bull River (Table 9). Diatoms accounted for >95% of total algal biovolume in the Bull River at all sites, but only 87% of total algal biovolume in the side channel. In the side channel, chlorophytes (5%), cryptophytes (2%) and dinoflagellates (3%) were more common, together accounting for 10% of the total algal biovolume.

A total of 27 diatom genera were found with those accounting for the most biovolume including Cymbella sp., Rossithidium sp., Achnanthidium sp., Fragilaria sp., Rhopalodia sp., Nitzschia sp., and Gomphonema sp. The genus Rhopalodia sp. was not present in the Bull River. The genus accounting for the most biovolume in the Bull River, Diatoma sp., was present in the side channel but at very low biovolume. Didymosphenia sp. was found in the Bull River, but not in the side channel. There were 18 green algal genera found in the side channel with Mougeotia sp. and Ankistrodesmus sp. accounting for the most biovolume. Species heterogeneity and richness were greater in the side channel compared to the Bull River.

There was no difference in mean periphyton PB across the four sites in the two years after the upgrade (one way ANOVA, $F_{3,4} = 0.89$. $R^2 = 0.4$, $p = 0.52$).

Table 9. Mean periphytic algal cell biovolume (±SD) by Division measured on artificial substrata at the three Bull River sites and in the side channel after the Aberfeldie upgrade (2010 and 2012).

Algal division or metric	N	Algal cell biovolume ($\mu m^3 \times 10^9/m^2$)			
		UBC1	MBT1	LBC2	SC
Diatoms	2	3667 ± 164	2887 ± 1021	3605 ± 1109	1959 ± 86
Chlorophyta	2	15.4 ± 21.8	47.5 ± 47.5	29.3 ± 27.8	113.2 ± 1.5
Cryptophyta	2	13.8 ± 19.5	37.5 ± 47	17 ± 24	47.6 ± 62.7
Cyanophyta	2	17.6 ± 4.2	24.3 ± 23.7	55.8 ± 70.9	45.6 ± 44.3
Euglenoids	2	13.4 ± 19.0	18.3 ± 25.9	9.2 ± 12.9	10.9 ± 15.4
Dinoflagellate	2	6.1 ± 8.6	22.9 ± 32.4	12.2 ± 17.3	76.5 ± 108.2
All Divisions	2	3733 ± 91	3025 ± 1197	3729 ± 1098	2253 ± 315
Richness (number of genera)	2	17.3 ± 3.3	18.8 ± 2.8	21.0 ± 0.7	25.0 ± 6.1
Simpson's Diversity Index (1-D)	2	0.75 ± 0.05	0.87 ± 0.02	0.82 ± 0.03	0.86 ± 0.03
Peak Periphyton Biomass (mg chl-a·m^{-2})	2	12.0 ± 2.1	21.3 ± 4.2	19.9 ± 12.1	20.4 ± 0.4

3.3 Invertebrate abundance in the diversion reach and side channel

The product of density and wetted area provided an estimate of total abundance of benthic invertebrates in the test segment of the diversion reach. This calculation was run before and after the upgrade in the non-pool wetted habitat of the test segment of the diversion reach.

Perrin and Bennett (2013a) showed there was no effect of the upgrade on total invertebrate density and chironomid density in the test segment of the diversion reach but there was a significant increase in EPT density. Therefore, the calculation of mean density of total invertebrates and chironomids was based on pooled measurements from all years except 2009 when an anomalous flow release could confound the density data (the 2009 flow anomaly is further explained by Perrin and Bennett 2013a). The result was that the same density value for total invertebrates and chironomids was used for describing densities before and after the upgrade as shown in Table 10. For EPT, the density data were separated by time period because there was a significant effect of the upgrade on the EPT density (Table 10).

Table 10. Mean density of benthic invertebrates (± SD) by site averaged over four years of sampling (pooled) or grouped by period before and after the Aberfeldie upgrade.

Metric	Test segment of the diversion reach before the upgrade	Test segment of the diversion reach after the upgrade	Test segment of the diversion reach using pooled data before and after the upgrade	Side Channel
N	2	2	4	2
Total benthic invertebrate density (number·m^{-2})	Not applicable	Not applicable	10793 ± 5991	4251 ± 2922
Chironomid density (number·m^{-2})	Not applicable	Not applicable	1930 ± 1350	1600 ± 937
EPT density (number·m^{-2})	6018 ± 451	9075 ± 5878	Not applicable	870 ± 839

Estimated invertebrate and chironomid abundance in the test segment declined by 19%. This change was entirely due to the 19% decline in wetted usable area because there was no significant change in density due to the upgrade (Table 11). EPT abundance in the test segment increased by 24% between the before and after periods due to a 19% decline in wetted area that was more than offset by a significant 50% increase in EPT density. These are maximum values because they assume the same use of wetted habitat by invertebrates throughout the wetted portion of the test segment of the diversion reach, regardless of substrate particle size.

Minimum invertebrate abundance was the product of density and wetted area not including boulder and bedrock (Table 12). Abundance differed between Tables 11 and 12 in relation to the difference in wetted usable area where boulders and bedrock were included in data showing maximum abundance and was not included in data showing minimum abundance.

Table 11 Maximum abundance of benthic invertebrates before and after the Aberfeldie upgrade in wetted areas of the test segment of the diversion reach.

Metric	N	In all wetted area of the test segment of the diversion reach before the upgrade	In all wetted area of the test segment of the diversion reach after the upgrade	Percent change
Median flow ($m^3 \cdot s^{-1}$)		6.3	2.0	-68%
Wetted non-pool habitat area (m^2)[a]		2494	2025	-19%
Maximum number of animals from all taxa ± SD[b]	4	$26.9 \times 10^6 \pm 14.9 \times 10^6$	$21.9 \times 10^6 \pm 12.1 \times 10^6$	-19%
Maximum number of chironomids ± SD[b]	4	$4.8 \times 10^6 \pm 3.4 \times 10^6$	$3.9 \times 10^6 \pm 2.7 \times 10^6$	-19%
Maximum number of EPT ± SD[b]	2	$15.0 \times 10^6 \pm 1.1 \times 10^6$	$18.4 \times 10^6 \pm 11.9 \times 10^6$	+23%

[a] From Table 5
[b]Standard deviations were calculated from replicate observations of the product of density and wetted area where one density observation was the mean density from the four samples in a given year. Since there was no effect of the upgrade on total invertebrate and chironomid density (Perrin and Bennett, 2013a), the two before and two after years were used for calculation of a mean resulting in n=4. Since there was an effect of an upgrade on EPT (Perrin and Bennett, 2013a), the `before` mean density was from 2005 and 2006 (n=2) and the after mean density was from 2010 and 2012 (n=2).

Table 12 Minimum abundance of benthic invertebrates before and after the Aberfeldie upgrade in wetted areas of the test segment of the diversion reach.

Metric	N	In wetted area of the test segment of the diversion reach not including boulder and bedrock before the upgrade	In wetted area of the test segment of the diversion reach not including boulder and bedrock after the upgrade	Percent change
Median flow in Diversion Reach ($m^3 \cdot s^{-1}$)		6.3	2.0	-68%
Minimum usable wetted non-pool area (m^2)[a]		1347	1094	-19%
Minimum number of animals from all taxa ± sd[c]	4	$14.5 \times 10^6 \pm 8.1 \times 10^6$	$11.8 \times 10^6 \pm 6.5 \times 10^6$	-19%
Minimum number of chironomids ± sd[c]	4	$2.6 \times 10^6 \pm 1.8 \times 10^6$	$2.1 \times 10^6 \pm 1.5 \times 10^6$	-19%
Minimum number of EPT ± sd[c]	2	$8.1 \times 10^6 \pm 0.6 \times 10^6$	$9.9 \times 10^6 \pm 6.4 \times 10^6$	+22%[b]

[a] From section 3.1
[b] Differs from the same cell in Table 11 due to rounding error
[c]Standard deviations were calculated from replicate observations of the product of density and wetted area where one density observation was the mean density from the four samples in a given year. Since there was no effect of the upgrade on total invertebrate and chironomid density (Perrin and Bennett, 2013a), the two before and two after years were used for calculation of a mean resulting in n=4. Since there was an effect of an upgrade on EPT (Perrin and Bennett, 2013a), the `before` mean density was from 2005 and 2006 (n=2) and the after mean density was from 2010 and 2012 (n=2).

Invertebrate abundance in the side channel is shown in Table 13. Maximum values are the product of total wetted area and mean density. Minimum values are the product of wetted area not including boulder habitat and mean density. The maximum and minimum values differ by approximately 5% because that was the proportion of wetted habitat in the side channel comprised of boulders. Chironomids represented 38% of total abundance, which is in contrast to 18% in the Bull River. The EPT represented 21% of total invertebrate abundance in the side channel, which is less than in the Bull River where up to 84% of total abundance was EPT.

Table 13 Range of benthic invertebrate abundance in the Aberfeldie side channel measured in 2010 and 2012.

Metric	N	In all wetted non-pool area of the side channel (maximum values)	In wetted non-pool area of the side channel not including boulders (minimum values)
Wetted non-pool area (m^2)		1564	1486
Number of animals from all taxa ± sd[a]	2	$6.6 \times 10^6 \pm 4.6 \times 10^6$	$6.3 \times 10^6 \pm 4.3 \times 10^6$
Number of chironomids ± sd[a]	2	$2.5 \times 10^6 \pm 1.5 \times 10^6$	$2.4 \times 10^6 \pm 1.4 \times 10^6$
Number of EPT ± sd[a]	2	$1.36 \times 10^6 \pm 1.3 \times 10^6$	$1.3 \times 10^6 \pm 1.2 \times 10^6$

[a] Standard deviations were calculated from replicate observations of the product of density and wetted area where one density observation was the mean density from the six samples in a given year. The data were from 2010 and 2012 (n=2).

Change in benthic invertebrate abundance (diversion reach plus side channel) resulting from the Aberfeldie upgrade is shown in Figures 13 (including all wetted areas) and 14 (not including bedrock and boulder). The decline in abundance of all invertebrates and chironomids in the diversion reach after the upgrade was entirely due to the 19% decline in wetted area in the test segment (Tables 11 and 12). Invertebrates in the side channel offset that loss, resulting in no net change in invertebrate abundance. This result was true for the combination of all invertebrates together and the chironomids. EPT abundance increased in the test segment of the diversion reach due to the upgrade as a result of a significant increase in density (Perrin and Bennett 2013a) that more than offset the loss of wetted area (Tables 11 and 12). Abundance of EPT in the side channel was only 7% (including all wetted habitat area) to 13% (not including boulder and bedrock) of that in the test segment, thus adding little to the net gain of EPT in the test segment.

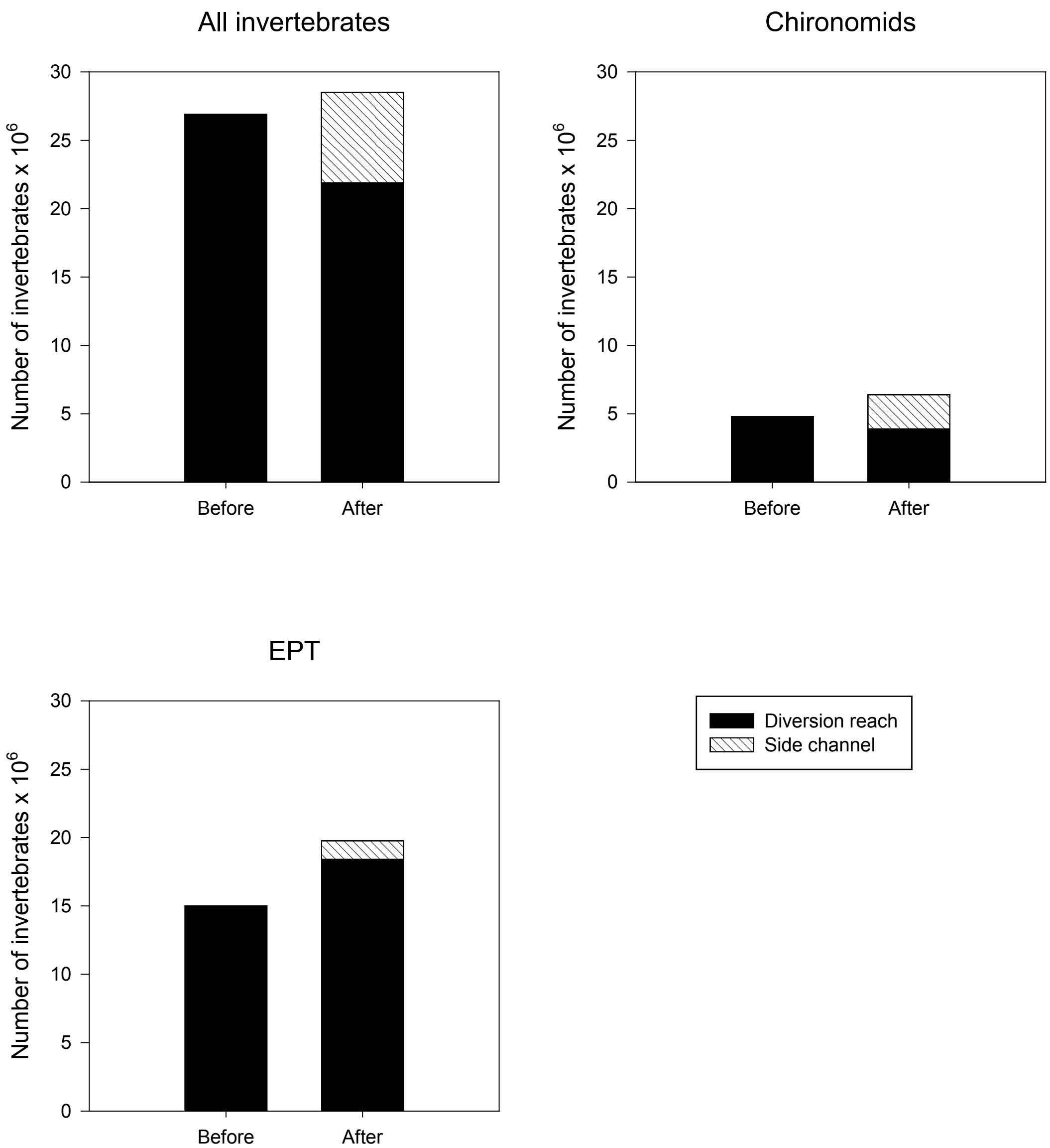


Figure 13. Cumulative estimated invertebrate abundance in the total wetted area of the test segment of the diversion reach and side channel before and after the Aberfeldie upgrade.

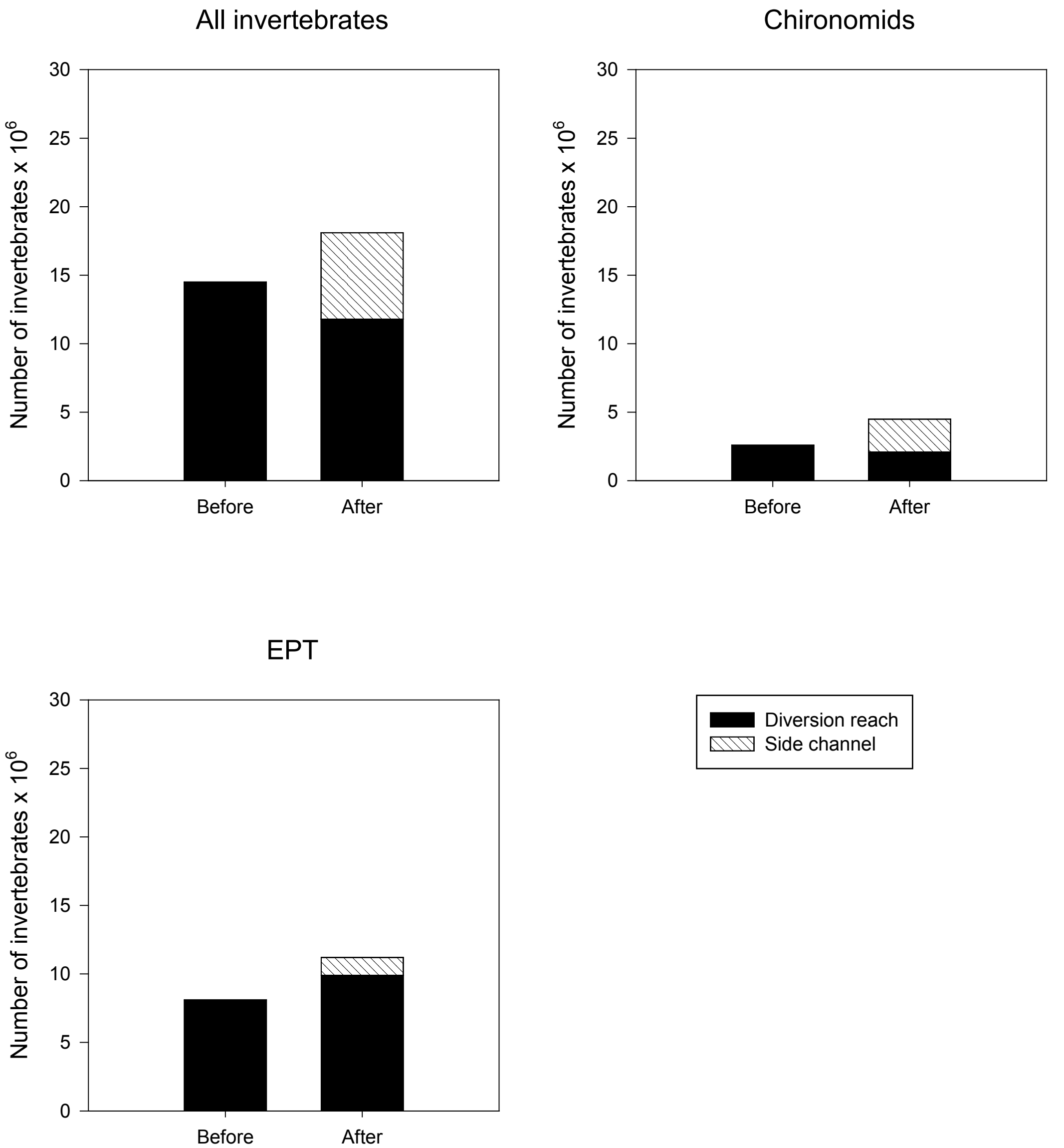


Figure 14. Cumulative estimated invertebrate abundance in the wetted area of the test segment of the diversion reach and side channel excluding boulder and bedrock habitat before and after the Aberfeldie upgrade.

3.4 Biotic abiotic matching

3.4.1 Quality of chemical analysis of water samples

Relative percent differences between replicate pairs of samples ranged between 2% and 38% (Table 14). 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 15). 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 14. 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 15. 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 (µg·L^{-1})*	Number of positive blanks (maximum possible is 10 (two samples per year over five years))	Average concentration in positive blanks (µg·L^{-1})	Average concentration in river samples (µg·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.

Measurement of habitat attributes at the time of invertebrate sampling showed that the side channel was chemically and hydrologically different from the Bull River (Table 16). Six of the environmental variables were used in the BEST analysis to examine biotic-abiotic matching among the river and side channel sites using data from 2010 and 2012. The selected subset of variables included dominant riparian vegetation class, embeddedness, median substrate particle size (D_{50} from Wolman pebble counts), dissolved oxygen (DO) concentration, turbidity, and mean flow.

Patterns in genus-level invertebrate assemblages were strongly matched with patterns in dissolved oxygen concentration (ρ =0.87, significance of test p=0.01). Various combinations of dissolved oxygen, average flow, median particle size, turbidity and embeddedness resulted in lower ρ values but were still highly correlated with the biota (BEST output, ρ ranged from 0.68 to 0.87). None of the solutions included the variable representing dominant riparian vegetation class. In all cases, low dissolved oxygen concentration in the side channel was the most important factor matched with differences in invertebrate assemblages between the side channel and river. While concentrations of dissolved oxygen were high in the river, they were low in the channel, often approaching 5 mg·L^{-1}, which is the lowest level required to support fish (CCME 1999). The side channel is groundwater fed and factors related to water source, including specific conductivity, dissolved oxygen, and turbidity were related to differences in invertebrate assemblages between the side channel and the river.

Table 16 Mean (± SD) values of habitat attributes at the three Bull River sites and in the side channel after (2010 and 2012) the Aberfeldie upgrade. Values in the side channel are shown as an average from sampling conducted along the upstream to downstream gradient of the channel. Environmental variables marked with an asterisk were included in the BEST analyses.

Environmental Variable	N	Bull River mainstem			Side Channel (SC)
		UBC1	MBT1	LBC2	
Dominant Riparian Vegetation Class*	2	Mixed forest	Mixed forest	Grass/herb	Deciduous forest
Water Temperature (°C)		9.5 ± 1.6	10.2 ± 1.7	10.1 ± 2.3	11.5 ± 1.6
Soluble reactive phosphorus ($\mu g \cdot L^{-1}$)	4	1.77 ± 0.75	1.89 ± 0.55	1.62 ± 0.81	3.53 ± 1.06
Median Pebble Size D_{50} (mm)*	2	180 Cobble	> 300 Boulder	180 Cobble	45 Gravel
Embeddedness*	2	0%	25%	25%	25 to 50%
Dissolved Oxygen ($mg \cdot L^{-1}$)*	15	10.6 ± 0.5	10.8 ± 0.6	10.6 ± 0.5	5.2 ± 1.2
Specific Conductivity ($\mu S \cdot cm^{-1}$)	15	302 ± 28	285 ± 39	292 ± 32	316 ± 42
Turbidity (NTU)*	15	2.0 ± 4.5	2.1 ± 4.2	2.6 ± 6.9	0.5 ± 0.4
pH	15	8.1 ± 0.5	8.4 ± 0.2	8.3 ± 0.1	7.9 ± 0.1
Water velocity ($m \cdot s^{-1}$)	15 site visits	0.96 ± 0.4 (n=53)	0.5 ± 0.39 (n=58)	0.79 ± 0.28 (n=60)	0.11 ± 0.12 (n=88)
Flow ($m^3 \cdot s^{-1}$)*	2 sampling periods	15.2 ± 6.4	3.3 ± 5.7	18.1 ± 7.6	0.012 ± 0.004

*Included in BEST analysis to determine the most important attribute contributing to dissimilarities of invertebrate assemblages between the Bull River and side channel.

4 DISCUSSION

There was an estimated 19% loss of benthic invertebrates in the test segment of the diversion reach with a reduction of flows from 6 $m^3 \cdot s^{-1}$ to 2 $m^3 \cdot s^{-1}$ due to the Aberfeldie upgrade. This change was entirely due to loss of 469 m^2 of wetted habitat area. The loss was offset by addition of invertebrates produced in the constructed side channel, resulting in no net loss of invertebrates from the Aberfeldie upgrade. Within the invertebrate community, there was no net loss of chironomids. EPT abundance increased in the test segment of the diversion reach due to the upgrade as a result of a significant increase in density that offset the loss of wetted area. Hence, there was no net loss of the EPT due to the upgrade. It had nothing to do with abundance in the side channel and everything to do with an increase in density in the test segment of the diversion reach due to the change in summer time flow.

These EPT are important because they are prey ingested by Bull trout, Westslope Cutthroat Trout (Schoby and Keeley 2011) and Mountain Whitefish (McPhail and Troffe 1998) that are present in the Bull River (Cope 2005). Bull trout tend to be more piscivorous than the other species but they also eat benthos, particularly larger invertebrates (Schoby and Keeley 2011). The EPT are the largest invertebrates found in the Bull River and are likely to be important prey for Bull trout as well as other fish species.

The statistical outcome of no effect of the upgrade on total invertebrates and chironomids was found despite an average 61% increase in the mean density of all invertebrates and an 88% increase in the mean density of chironomids between periods before and after the upgrade in the test segment of the diversion reach (Perrin and Bennett 2013a). The power of the tests was low (8 to 11%), meaning there was little chance of detecting an effect of the upgrade on total invertebrates and chironomids if it was present. The low power was due to few replicate years of sampling. If the increases in mean density of total invertebrates and chironomids were significant, they would offset losses due to the decline in wetted habitat area and result in a net gain in abundance due to the upgrade. In addition, given the large size of EPT individuals compared to the other taxa, the effect of change in flow on biomass of total invertebrates may be significant due to a potential increase in EPT biomass related to change in flow in the test segment of the diversion reach. Hence, the outcome of a net 19% decline of numbers of all invertebrates and the chironomids related to the change in wetted area is conservative. Actual change may be greater and potentially similar to that found for the EPT portion of the community in which abundance increased 23% in the test segment of the diversion reach due to the upgrade.

The total wetted area in the test segment of the diversion reach after the upgrade at the prescribed minimum flow of 2 $m^3 \cdot s^{-1}$ is considered accurate because it was directly measured using standard survey techniques from multiple transects (Perrin and Canning, 2010). In contrast the values of wetted area at pre-upgrade flows were modeled using calibration from only two ground transects, which had error (Table 1; Section 2.2). While there was little choice but to develop a model to predict wetted area at the pre-upgrade median flow of 6 $m^3 \cdot s^{-1}$, the prediction was based on extrapolation of the model beyond the range of data upon which it was based and it included only three data points. Certainly, error was present using this approach (Quinn and Keogh 2002). This dilemma is the same as was encountered by Cope (2005) for calculation of wetted areas at flows lower than were measured. It can, however, be resolved. A wetted area survey should be done using survey techniques incorporating at least nine transects as was done by Perrin and Canning (2010) in the test segment of the diversion reach. A resulting measurement of wetted area can then be used to recalculate invertebrate abundance in the test segment of the diversion reach at the pre-upgrade median flow of 6 $m^3 \cdot s^{-1}$.

A management question asked whether there is a summer minimum flow different from the 2 $m^3 \cdot s^{-1}$ in the test segment of the diversion reach that, in combination with invertebrates from the side channel, could achieve no-net-loss of productive capacity. Presumably this alternate flow would be lower than 2 $m^3 \cdot s^{-1}$ given that lower flow in the diversion reach would result in more water flowing to the powerhouse. There was an increase in abundance of the EPT in the test segment with a reduction in median flow from 6 $m^3 \cdot s^{-1}$ to 2 $m^3 \cdot s^{-1}$, a 67% decline in flow. Responses of invertebrate densities to flows lower than 2 $m^3 \cdot s^{-1}$ were not tested in this study but some insight about potential change can be gained from other studies. In their review, 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) 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 possible confounding by various habitat attributes, an increase in abundance of EPT with a decline in flow was found in the Bull River. It may be related to tolerances of EPT to changes in hydraulic stress (e.g., Rempel et al. 1999) at different flows. Highest densities occurred at low flows when water depths and velocities would be lower than at high flows. Rempel et al. (2000) showed that benthos, EPT in particular, are sensitive to hydraulic gradients, with many occurring in highest density in shallow water where hydraulic stress is lowest. This sensitivity may be present in the Bull River. A shift to lower flows in August and September after the upgrade at Aberfeldie may have caused hydraulic stress to decline, potentially favouring EPTs. Given the variation in findings from other studies about change in invertebrate abundance to declining flows, it is unknown how far flows can decline before this advantage of lower hydraulic stress is overcome by other factors that cause a decline in invertebrate abundance. As a result, the extent of flow reduction below 2 $m^3 \cdot s^{-1}$ in the test segment before EPT abundance declines is presently unknown and cannot be determined with existing data.

An argument might be that losses of EPT and thus important fish food organisms from the diversion reach do not matter because the side channel can be used to offset those losses at flows lower than 2 $m^3 \cdot s^{-1}$. Unfortunately the present structure and capability of the channel cannot ensure compensation. EPT abundance in the side channel was only 7% of that produced in the test segment of the diversion reach and the channel favoured small sized chironomids, black fly larvae (Simulium sp.), naidid worms, and ostracods. This community was entirely different from that in the Bull River and it lacked the EPTs that are known to be important food for the fish species found in the Bull River (Schoby and Keeley 2011, McPhail and Troffe 1998). The side channel community was correlated with low dissolved oxygen (DO) concentrations that approached levels known to limit use of the habitat by aquatic organisms. Half of the DO concentrations were ≤5 $mg \cdot L^{-1}$ that is too low to support salmonids (CCME 1999, BC MOE 1997). The minimum DO concentration for protection of cold water aquatic life is 9.5 $mg \cdot L^{-1}$, with a lower concentration of 6.5 $mg \cdot L^{-1}$ for protection of cold water early life. Similarly, instantaneous minimum DO concentration for protection of aquatic life is 9 $mg \cdot L^{-1}$ for buried embryos/alevins, and 5 $mg \cdot L^{-1}$ for all other life stages.

Conditions could be improved in the side channel to make it more suitable to support EPTs, that are common in the Bull River and are known food for rearing char and salmonids. Surface water could be supplied via an intake installed in the Bull River. This one action would increase flow rates and invertebrate recruitment. Surface water diversions from the river mainstem would provide oxygen concentrations near saturation that would alleviate the present low DO concentrations and provide conditions similar to those of the Bull River because of a direct link to the same water supply. This direct link to the same water supply would be expected to support a periphyton community similar to that in the Bull River that in turn would support invertebrates typical of the Bull River, including the EPT. While this change in water supply intuitively is attractive, it should not be done if it reduces the wetted habitat area of the Bull River mainstem. There is little point in simply redirecting water down the side channel if it does not change area of suitable habitat to support benthos and fish. Calculations are recommended to be run to determine if such diversions would result in a net increase in wetted area of usable habitat before actions are taken to change the water supply. Until these calculations are done and show that a change in water supply is warranted and commensurate improvements are made to the side channel, no summertime reductions in flow in the diversion reach should be implemented. Otherwise a net loss of important fish food organisms from the project may occur.

Challenges with successful development of constructed side channels in rivers are not uncommon. Jones et al. (2003) showed that a 3.4 km constructed channel in the Northwest Territories did not meet expectations for supporting Arctic grayling because of lack of organic matter to support a food web. Further study showed that ten or more years could be required to establish the instream and riparian vegetation needed for organic matter recruitment that is similar to natural channels (Jones et al. 2008). In contrast, a constructed stream in Newfoundland was successfully colonized by invertebrates and the community was similar to natural channels within three years (Gabriel et al. 2010). Similar to the Northwest Territories example, the invertebrate composition was linked to the source of flow and organic matter input from riparian vegetation. In another example, the permanent wetting of a 1.2 km channel using various flow control structures was successful in creating stream habitat in Newfoundland following three years of development (Scruton et al. 2005). Again flow supply and control was critical to success. Given these examples, successful compensation for potential loss of biological production in the Bull River at flows lower than 2 $m^3 \cdot s^{-1}$ in summer might be achieved in the Aberfeldie side channel but it requires concerted effort to supply adequate flow from natural surface sources. This attention to hydrology is needed to achieve effective biological production to support fish communities. If changes are made to the channel, monitoring of the channel hydrology, morphology, water chemistry, and biological communities will be required to properly assess channel effectiveness.

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