

Ecosystem level assessment of environmentally based flow restrictions for maintaining ecosystem integrity: a comparison of a modified peaking versus unaltered river

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ABSTRACT

Although dams have impounded the majority of the world's altered watercourses, there is a growing awareness of the importance of mitigating or reversing some of the negative effects on aquatic ecosystems and the related services they provide. We used an ecosystem approach, including detailed studies on hydrology, geomorphology, invertebrates, fish, and food web dynamics on a river altered by waterpower production and a natural flowing river to assess system responses to a change in the altered flow regime (specifically the ramping rate or rate of change of flow). Although there was significant alteration in the flow and sediment regimes under the original restricted ramping rate regime, differences in many biotic variables in the two rivers were not significant including total invertebrate abundance and diversity, fish biomass, fish condition, and food web length. However, significant differences in the abundance and distribution of some sensitive invertebrate taxa and fish diversity were observed between the altered and natural flowing rivers as was the energy base of the food web, measured with stable isotopes. The altered river had lower overall abundance of Odonata, Ephemeroptera and Plecoptera, and Diptera, Trichoptera, Ephemeroptera, and Coleoptera increase in abundance towards the deeper and higher velocity thalweg. On average, $\delta^{13}\text{C}$ values were lighter in altered sites compared to unaltered sites, likely due to carbon export from the upstream reservoir. Results will inform Canadian federal and provincial policy concerning the efficacy of ramping rate restrictions as a tool to mitigate the environmental impacts associated with peaking waterpower dam operations. Copyright © 2010 John Wiley & Sons, Ltd. and Crown in the right of Canada

KEY WORDS ecosystem; assessment; river; flow; restrictions; modified peaking

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INTRODUCTION

An estimated 60% of the world's watercourses have been subjected to hydrologic alteration, with the most prevalent cause being the impoundment by dams (Tharme, 2003). Many dams are built for hydroelectric generation, which in Canada comprises approximately 63% of the total power supply (Canadian Electricity Association (CEA) 2009). Both the size and the operation of dams vary, with run-of-the-river generally considered most benign and full peaking most damaging (Clarke *et al.*, 2008). Full peaking results in high discharge coincident with high electricity demand; when demand is low, flow

decreases and ranges from only dam 'leakage' to an imposed minimum discharge (Clarke *et al.*, 2008). Relative to natural flow regimes, peaking operations greatly alter the magnitude, duration, timing, frequency, and rate of change of flow, which can lead to altered temperature regimes and geomorphology, changes in biogeochemical cycling, reduced habitat diversity, organism physiological stress, and the reduced abundance, diversity and productivity of biota (Cushman, 1985; Richter *et al.*, 1997; Bunn and Arthington, 2002; Sabater, 2008).

In recent years, however, resource managers and regulatory authorities have begun to impose constraints on the operation of peaking facilities to move them in the direction of a natural flow regime, the underlying tenet of 'the Natural Flow Paradigm' (NFP; Poff *et al.*, 1997). The NFP is based on the contention that organisms have adapted to the variations inherent in natural flow

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regimes and that the ecosystem integrity of a river is dependent on maintaining such variability (Poff *et al.*, 1997; Richter *et al.*, 2003). Environmental flows, or flows prescribed for the benefit of river ecosystem health, historically focussed on minimum flow requirements but have evolved to emphasize all ecologically important elements of the natural flow regime (high-to-low flows) and their associated characteristics, e.g. frequency, duration, timing, and rate of change. A wide variety of objectives and methods are used to prescribe environmental flows, ranging from expert opinion to detailed hydraulic and habitat modelling, and no one technique is necessarily considered superior (Acreman and Dunbar, 2004). The belief is that integrating ecologically important components of the natural flow regime into an altered flow regime will mitigate some of the documented negative environmental impacts (Enders *et al.*, 2009).

In general, waterpower facilities capture water during high flows for more controlled release when electricity is needed. In the case of peaking facilities, water capture and control drastically increases the frequency and rate of change of flows relative to a natural system (Clarke *et al.*, 2008). For example, a typical pattern for a peaking facility would be maximum flow during daytime peak energy demand periods and decreased flow at night and weekends when demand is often lower. Thus, for peaking facilities, one way to mitigate the degree of alteration to the flow regime is to constrain the rate of change of flow (or ramping rate) allowed. While resource managers believe ramping rate restrictions provide environmental benefits by reducing the negative effects typically associated with dam operation, there is a paucity of scientific information on the resulting ecological benefits.

Ramping rate restrictions are assumed to impede the efficiency and profitability of electricity generation. Therefore, it is important to assess the ecological benefits and risks associated with changes to ramping rates and to develop the scientific information necessary for the design of integrated management tools that address both ecosystem health and energy efficiency. To that end, collaborative research involving scientists, regulatory agencies, and industrial partners was undertaken to test whether quantifiable ecological benefits of constraining ramping rates in a peaking flow regime are realized. This manuscript presents a thorough evaluation of ecological conditions that existed during the baseline phase of a long-term ramping rate experiment that is ongoing and evaluates the benefit of peaking restrictions relative to an unaltered reference river. Here, we report results from a comparative analysis of hydrology, fluvial geomorphology, invertebrates, fish, and food web metrics between a river altered by a hydroelectric dam operating in a restricted peaking mode and an unaltered reference river.

STUDY DESIGN

This study is a collaboration between Fisheries and Oceans Canada, the Ontario Ministry of Natural Re-

sources, Brookfield Renewable Power Limited, and the University of Waterloo. Two rivers located on the boreal shield were compared for this study: the experimental Magpie River, Wawa, Ontario (Magpie 48°4'N; 84°44'W), with a flow regime altered by the Steephill Falls waterpower facility (WPF) and the Batchawana River (Batchawana -47°0'N; 84°30'W) with a natural flow regime located 60 km north of Sault Ste. Marie, Ontario (Figure 1a). The Magpie drains an area of ~1930 km² with a mean annual flow (MAF) of ~27 m³ s⁻¹ observed over a 51-year record preceding the initial operation of waterpower facilities in 1991. The watershed is predominantly bedrock with glaciofluvial deposits. The Batchawana drains an area of ~1233 km² and has a historical MAF of ~22 m³ s⁻¹ (1969–2007). The Batchawana watershed is predominantly bedrock with less glaciofluvial deposits compared to the Magpie but more organic deposits, particularly lower in the watershed. As a result peak flows in the Batchawana are similar to those of the Magpie natural flow regime, but with steeper rising and falling limbs associated with snowmelt and rain events, and lower baseflow between events. Batchawana also exhibits earlier initiation of springmelt, owing to its more southerly location. Physical properties of a suite of potential reference systems were thoroughly examined via a Geographic Information System (GIS) analysis of landscape characteristics (Metcalf *et al.*, 2001), and the Batchawana River was considered most suitable in terms of comparable fish communities, size of river, slope, geology, and availability of historical flow data.

The WPF was commissioned in 1989 and was required to include minimum flow and ramping rate restrictions as part of its operating requirements since that time through the duration of results reported here. The restrictions were put in place by the Ontario Ministry of Natural Resources as part of the Waterpower Lease agreement as they were considered protective of the valued brook trout population in the river. The peaking WPF draws water from a depth of 10 m in the reservoir (total reservoir depth of 30 m) or approximately a metalimnetic (cool water) draw. The maximum flow passable through the turbines is 44 m³ s⁻¹ and the minimum discharge is restricted to 7.5 m³ s⁻¹. If reservoir levels exceed the rule curve for the reservoir during spring freshet, additional water is spilled over and above the 44 m³ s⁻¹, but otherwise this maximum is not exceeded. If the WPF ceased operation for any reason, spill gates automatically opened to release the minimum 7.5 m³ s⁻¹. From the date of commissioning through 2004, ramping rates were restricted to ≤1 m³ s⁻¹ h⁻¹ from 10 October to 15 November, and 2 m³ s⁻¹ h⁻¹ from 16 November until spring freshet (early May). From May until early October, restrictions included an increase or decrease of 25% of the previous hour's flow.

For purposes of stratified random sampling, the river was divided into valley segments (VSs), defined as areas of the river that were physically distinct in terms of flow, depth, surficial geology or at the confluence

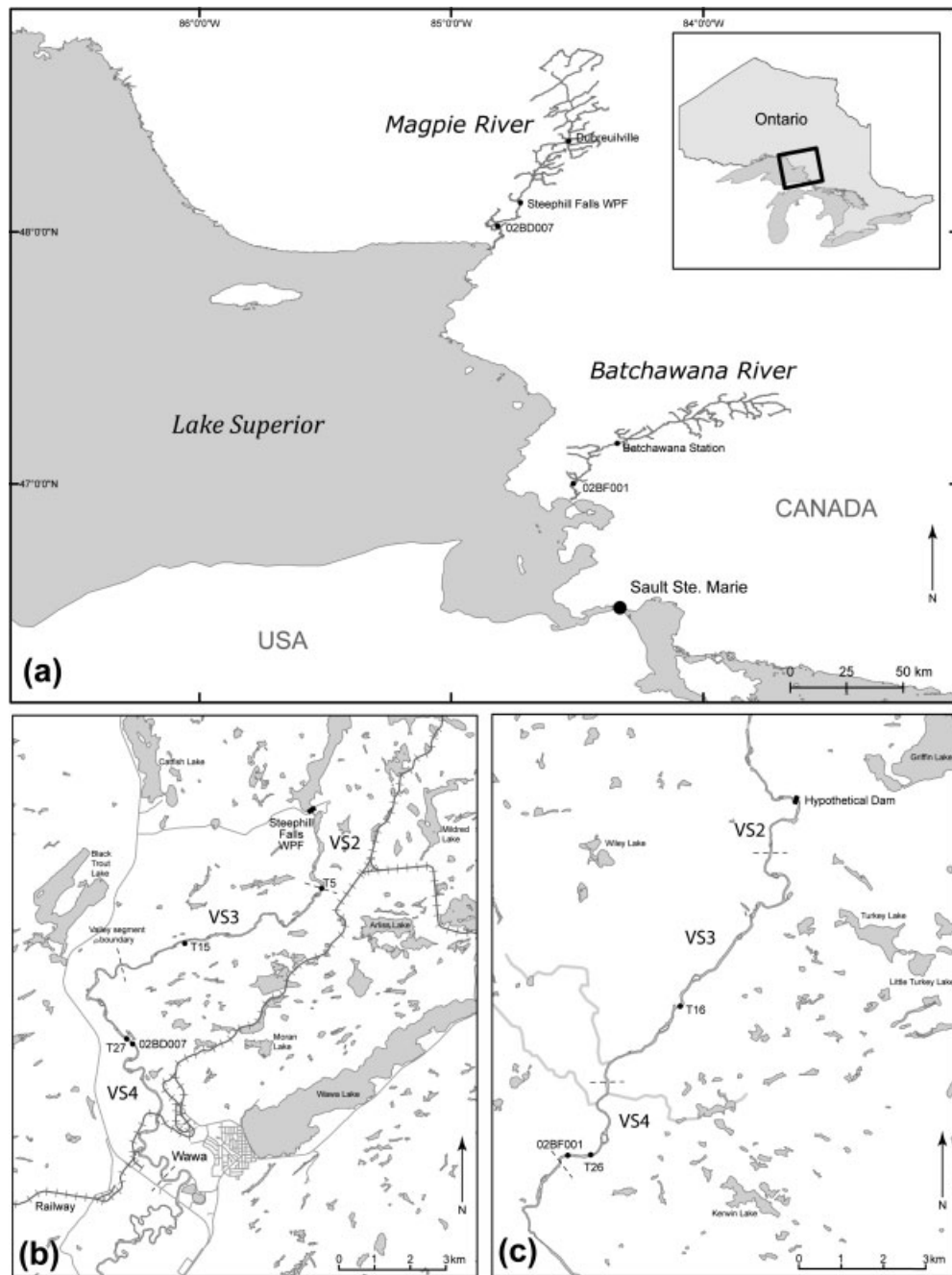


Figure 1. Location of the (a) Magpie (altered) and Batchawana (natural) Rivers in Ontario, Canada, and detailed sections of the study reaches on the (b) Magpie and (c) Batchawana, showing valley segments (VS) and some transects (T) for spatial reference. Steephill Falls WPF, select transect locations (e.g. T27), and Water Survey of Canada streamgauges (02BD007, 02BF001) are shown for reference.

of a major tributary. On the Magpie, VS1 is located near Dubreuville, and on the Batchawana VS1 is near Batchawana Station (Figure 1a). Valley segment 1, 25 km above the dam, was outside the zone of influence of the reservoir and was included as an internal control in the sample design (identified as MR-Upper). On the Magpie, the most intensively sampled VSs were those considered most vulnerable to ramping due to their riffle-run-pool riverine nature (VS3 and VS4). The spatial and sampling coverage of VSs was similar on the Batchawana. To facilitate sampling and spatial reference, geo-referenced transect markers (T's) were placed along the riverbanks at 500-m intervals from the dam (e.g. Magpie River: MR-T1

is located 500 m downstream of Steephill Falls WPF) or from a fixed location on the Batchawana River (e.g. BR-T1—each transect is 500 m apart).

METHODS

Hydrology

Year-round discharge measurements at Steephill Falls WPF and Water Survey of Canada (WSC) stream-gauge 02BD007 located 13.5 km downstream on the Magpie (MR-T27) and stream-gauge 02BF001 on the Batchawana (BR-T26) were supplemented with other

instream measurements of water level at sites distributed longitudinally on both rivers. Measurement sites were located in relatively straight sections with well-defined banks and stable cross-sections. This included two sites located 2.5 km (MR-T5) and 7.5 km (MR-T15) downstream of the Steephill Falls WPF and at MR-Upper. MR-T5 was located downstream of the three pools attenuating changes in flow downstream of the WPF and upstream of the Lena Creek confluence to determine the 'effective' ramping rate (Figure 1b). The 'effective' ramping rate was distinguished from the 'operational' ramping rate, measured at Steephill Falls WPF, to address potential influences of the pools and ensure transferability of results to other systems without natural attenuation. Another site (BR-Upper) was located 35 km upstream of streamgauge 02BF001 on the Batchawana near Batchawana Station to cover a similar longitudinal extent. Historical records from WSC streamgauge 02BD003 located near the confluence of the Michipicoten River were prorated for the Steephill Falls WPF to provide an indication of the Magpie's natural flow regime before the alteration.

Vented pressure transducers (Keller 173) were installed at MR-T5, T15, Upper, and BR-Upper to monitor water level while the WSC supplied discharge data from the streamgauges at MR-T27 and BR-T26. Sensors were attached to cinder blocks and cabled to Campbell Scientific CR10x data loggers located in stainless steel housings on shore. Instrumentation was deployed following springmelt (early June) and removed in October. Hourly data were acquired from WSC streamgauges 02BF001, 02BD007, and Steephill Falls WPF while all other measurements were logged at 15-min intervals. Rating curves were established at MR-T5, T15, Upper, and BR-Upper using the depth-area velocity method for measuring discharge (Maidment, 1993). Velocity profiles were obtained using a Marsh-McBirney Model 2000 Flowmeter, SonTek FlowTracker, or SonTek RiverSurveyor ACDP.

Fluvial geomorphology and channel change potential of the Magpie

Cross-channel transects were surveyed on both rivers to investigate associations between channel morphology and altered riverine flows. The transects were surveyed using a level, stadia rod, and tripod, and channel bed slope and surface water slope were also recorded at each cross-section. Wolman (1954) pebble counts at each transect were used to estimate bed sediment size distributions. Bankfull stage and width, floodprone width, and entrenchment ratio were also measured. The cross-sections were characterized using the Rosgen (1996) classification system. A simple method was used to determine flow depth to initiate bed sediment transport, using the critical shear stress equation for motion for a given bed sediment size (in Dubinski and Wohl, 2007, adapted from Komar, 1987). The median grain size (D_{50}) is commonly used in sediment transport calculations; however, Ryan *et al.* (2005) indicated that the movement of the D_{16} to D_{25} grains from the bed surface corresponded with the

onset of channel maintenance flows; these grain sizes are used in the sediment transport analysis. Twenty-four years of daily flow data (1966–1989) were examined from Water Survey of Canada gauge 02BD003 (Magpie River near Michipicoten) to determine how often flows occurred to mobilize bed material at MR-T27. The flow data were prorated using drainage basin area from the WSC gauge site to the MR-T27 site.

Benthic invertebrates

To fully assess the invertebrate community spatially across impact zones (wet–dry vs permanently wet), two sampling methods were used. To assess invertebrate abundance and diversity in the varial zone (the area of streambed that is wet and dried in relation to power generation Korman and Campana, 2009), benthic invertebrates were sampled mid-July of each year with a Surber sampler (0.09 m², 500 µm mesh) and subsequently preserved in 70% ethanol. Samples ($n = 6–10$) were equally spaced along cross-sections leading from the wetted shallows to progressively deeper waters near the thalweg. Cross-sections were selected in riffles that gradually increased in water depth and showed large changes in wetted width in relation to changes in flow. Effort was made to sample similar habitat (e.g. substrate, exposure) to reduce variability from other potentially confounding factors. The Magpie had three cross-sections at MR-T5, T9, and T15. Similarly, three cross-sections were spaced 5 km apart in the Batchawana at BR-T16, T23, and T27. Water depth (cm), velocity (m s⁻¹), and substrate size composition (Wentworth scale) were recorded at each sampling point. In the laboratory, invertebrates were identified to genus, with the exception of Nematoda, Turbellaria, and terrestrial invertebrates, which were typically identified to Family or Order.

Correspondence analysis (CA) was used to ordinate associations among the invertebrate taxa (abundances) in the studied rivers (using BiPlot, Excel add-in; Lipkovich and Smith, 2002). This indirect gradient analysis procedure provides information about the similarity between rivers based on the abundance of taxa present in each community (Legendre and Legendre, 1998). Abundances of invertebrates were $\log(x + 1)$ transformed to down weight the influence of both dominant and rare taxa (Keller *et al.*, 2002).

To assess invertebrate abundance and diversity in the permanently wetted zone of the rivers (thalweg), six riffle sites were chosen on each of the Magpie and the Batchawana, one at MR-Upper and five downstream at MR-T5, T6, T12, T19, and T21. The sites on the Batchawana were selected to cover a similar longitudinal extent. Each year and at each site, five rock bags (5 cm mesh netting, 46 cm length) were filled with rocks of representative size found along the shoreline at the site of placement, until each reached a weight of 7 kg (± 0.5 kg). The bags were left in the river for a period of approximately 60 days (June–August, 2002–2004), a sufficient length of time for full colonization to reach

fluctuating taxa richness, abundance, and biomass (Mason *et al.*, 1973; Shaw and Minshall, 1980). Once bags were retrieved, the rocks were cleaned and all invertebrates and debris were preserved in 70% ethanol. The entire sample was subsampled for taxonomic identification to family and enumeration using guidelines provided by Environment Canada's National Water Research Institute for area-based sieve splitting National Water Research Institute (NWRI), 2002 which was originally described by Cuffney *et al.* (1993).

The number of invertebrate families was used to calculate taxonomic richness, probability of interspecific encounter (PIE; Hurlburt, 1971) and %Ephemeroptera–Plecoptera–Trichoptera (%EPT). PIE is an unbiased evenness measure that evaluates the chance that two individuals drawn at random from a population represent different species. PIE was selected over other diversity indices because it provides a statistically and biologically understandable probability, unlike more traditional diversity measures (Gottelli and Graves, 1996). The %EPT calculations were completed by summing the number of individuals within the families of EPT and dividing by the total number of individuals in all invertebrate families found in the samples. Both indices were calculated using formula generated in Excel spreadsheets and other rock bag statistics were conducted using Statistica (6.1, StatSoft Inc.) software.

Fish

Backpack electrofishing was used to examine relative fish population abundance, biomass, growth, species richness, diversity, and community composition on both rivers. Electrofishing was conducted in July of each year on the Batchawana River and in August of each year in the Magpie River. A stratified random sampling design by valley segment (VS1, VS3, and VS4) was used to ensure spatial coverage of the rivers, yet maintain statistical integrity. At each site, all areas ≤ 60 cm in depth were sampled using backpack electrofishing covering twelve 100-m shoreline segments on the rivers. These segments were sampled according to habitat type (fast or slow), and shocking methods were standardized by power (W), rate ($s\ m^{-2}$), area (m^2), and time (s). Fish were identified to species and enumerated. A random sub-sample of the captured fish was preserved for accurate length and weight data.

Fish biomass per unit area ($g\ 100\ m^{-2}$) was calculated for each sampling site. Data were $\log(x + 1)$ transformed as necessary to meet assumptions of normality and homogeneity of variance. Fish statistics were calculated using Statistica (6.1, StatSoft Inc.) software. For each river, a 3-way ANOVA was completed using year (2002–2004), habitat type (fast or slow), and valley segment (VS1–VS3–VS4) as the independent variables. When differences were found, Tukey's *post hoc* honestly significantly different (HSD) test was applied. Diversity was measured using PIE in a 3-way ANOVA (river–year–habitat), excluding VS1 and blending VS3

and VS4 (previously tested with no difference) to test for the effect of the dam on the downstream fish community. VS1 (Magpie—Upper) was excluded as it was outside the WPF zone of influence, and here, we are interested in testing for an impact of the dam. A CA was conducted on fish counts by year and VS ($\log(x + 1)$ transformed to down weight the influence of dominant taxa) to ordinate associations of fish communities in the rivers. For long-nose dace (*Rhinichthys cataractae*), troutperch (*Percopsis omiscomaycus*), brook trout (*Salvelinus fontinalis*), and slimy sculpin (*Cottus cognatus*), length–weight relationships were compared between rivers using ANCOVA on a randomly selected subset of 100 individuals (stratified by length) when the sample size was greater than 100.

Stable isotopes

Samples for carbon and nitrogen stable isotope (SI) composition were collected in both Magpie and Batchawana twice a year (spring and summer) in 2003 and 2004 to determine main food source and consumers' trophic position. Aquatic vegetation samples consisted of representative macrophytes and algal taxa (mostly *Potamogeton richardsonii*, *Ceratophyllum demersum*, *Vallisneria americana*, and filamentous algae) and were sorted as emergent, submergent, and algal taxa. Aquatic invertebrates were collected using a surber sampler placed downstream of areas vigorously disturbed by kicking. At shore, organisms were sorted into major taxonomic groupings (stoneflies, caddisflies, damselflies, dragonflies, mayflies, water striders, clams, and snails). Fish were captured by electrofishing and sorted according to species. A detailed list of invertebrates and fish species sampled from these rivers is presented in Marty *et al.* (2009).

In the laboratory, plants, invertebrates, and dorsal fish muscle tissue were dried ($50^\circ C$) and grounded using either a mortar or a ball mill grinder. Samples were not acidified or lipid extracted because of circumneutral pH found in both rivers and the uniform C/N ratios found in all animal taxa. Three samples from each sample grouping were analysed with the exception of fish, where up to ten individuals covering the size range of each species were analysed. SI signatures were measured at University of Waterloo, Environmental Isotope Laboratory, using a Thermo Finnigan Mat Delta Plus mass spectrophotometer coupled to a Carlo Erba NA1500 analyzer. Results are given using the standard δ notation, expressed in parts per thousand (‰). A secondary standard (cellulose) of known relation to the international standards of Pee Dee Belemnite (Craig, 1957) and atmospheric nitrogen (Mariotti, 1983) was used as a reference material. Average precision on SI measurements was 0.11 and 0.08‰ for carbon and nitrogen signatures, respectively.

The large number of samples collected in this study (over 1000) allowed for the simultaneous examination of several variables having possible significant impacts on isotope signature variance. A restricted maximum likelihood analysis of variance was applied to account for

unbalanced data sets and detect significant differences in the isotope signatures of taxonomic groupings obtained between years, seasons, and site type (altered or natural flow). All SI result statistics were conducted using JMP (7.0.1 SAS Institute).

RESULTS

The suitability of the Batchawana River as an appropriate control for the Magpie was assessed on the basis of its proximity to the Magpie (for climate control), with a natural flow regime similar to the Magpie pre-alteration, a historical streamflow record, a natural barrier to Lake Superior (falls), and comparable channel lengths, widths, and fish communities. Observed streamflow on the Batchawana for the study years was similar to Magpie's flow regime pattern prior to alteration (Figure 2a). Water chemistry samples also confirmed the similarity of the study rivers, with parameter values being slightly higher in the Magpie: pH (2003 values, mean \pm SD: 7.6 ± 0.1 , Magpie; 7.3 ± 0.3 , Batchawana), conductivity (92 ± 17 $\mu\text{mhos cm}^{-1}$, Magpie; 38 ± 11 $\mu\text{mhos cm}^{-1}$, Batchawana), total phosphorus (8.0 ± 1.6 $\mu\text{g l}^{-1}$, Magpie; 7.0 ± 1.9 $\mu\text{g l}^{-1}$, Batchawana), and chlorophyll *a* (1.7 ± 0.5 $\mu\text{g l}^{-1}$, Magpie; 0.86 ± 0.6 $\mu\text{g l}^{-1}$, Batchawana).

Hydrology

The altered streamflow pattern observed at Steephill Falls is very different from the natural flow regime observed on the Magpie prior to waterpower development (Figure 2b),

partly the result of technological and regulatory constraints [e.g. maximum turbine flow ($45 \text{ m}^3 \text{ s}^{-1}$) and low flow requirement ($7.5 \text{ m}^3 \text{ s}^{-1}$) and ramping restrictions, respectively]. Ramping restrictions were particularly important for influencing streamflow pattern by prohibiting the achievement of a daily peaking cycle from the low flow requirement to maximum turbine flow. Consequently, when water supply was not limited the flow pattern would typically include minimum required flows on weekends and a saw-toothed pattern through the weekdays perched on an elevated low flow (Figure 2b (i)). The discharge pattern ensured that the next daily maximum flow could be achieved under the ramping restrictions. The pattern was dampened when water supply was less (Figure 2b (ii)) and curtailed severely under even drier conditions (Figure 2b (iii)), given the volume of water required to maintain the elevated peaking pattern.

Flow alteration by the Steephill Falls WPF resulted in a range of observed flows that were dampened compared to a natural flow regime (Figure 3a). The maximum turbine flow and minimum flow constraints also influenced the shape of the flow duration curve (FDC), evidenced by the flattened portions of the curve at higher and lower magnitude flows, respectively (Figure 3b). The remaining time was largely comprised of transitional flows or maximum flows less than the maximum turbine flow. Rates-of-change of flow were higher on the Magpie compared to the Batchawana and the proportion of time associated with ramping (~ 22 – 42%) was significantly greater than the proportion of time associated with the rising and falling limbs of naturally occurring streamflow events on the Batchawana ($\sim 10\%$) (Figure 4). The influence of the

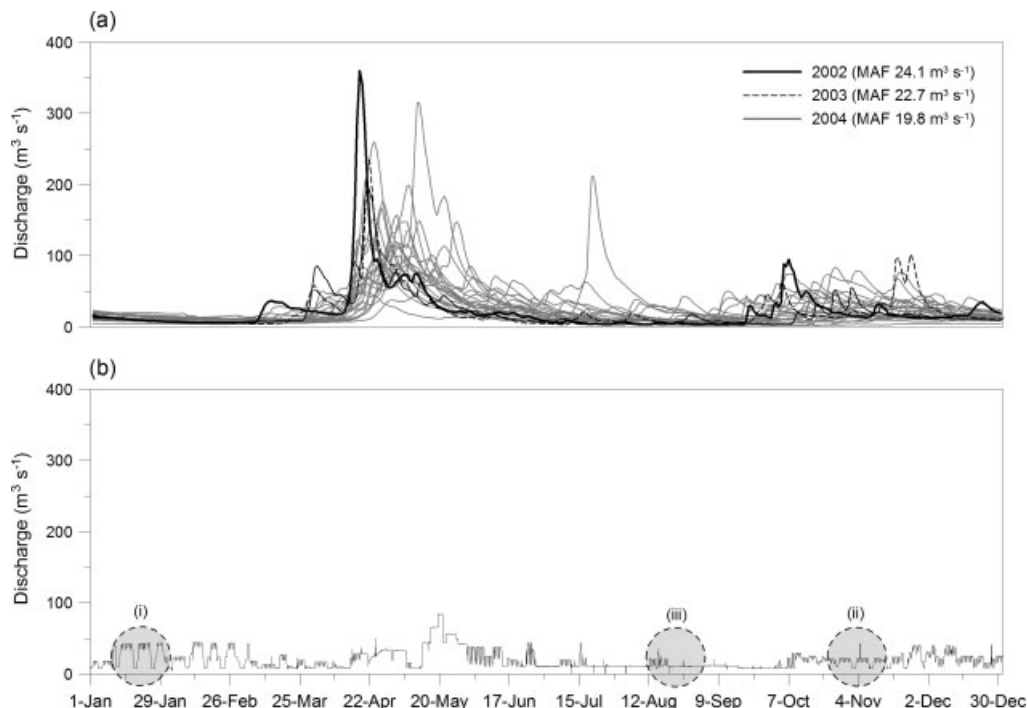


Figure 2. (a) Natural flow regimes recorded for the period of record at streamgauge 02 DB003 on the Magpie and prorated for the Steephill Falls WPF (grey lines) and for streamgauge 02BF001 on the Batchawana for 2002–2004 (black lines) compared to (b) the altered flow regime at the Steephill Falls WPF in 2003. Circled portions of the altered flow regime indicate typical flow patterns when (i) water supply was not limited, (ii) less water was available, and (iii) during drying conditions.

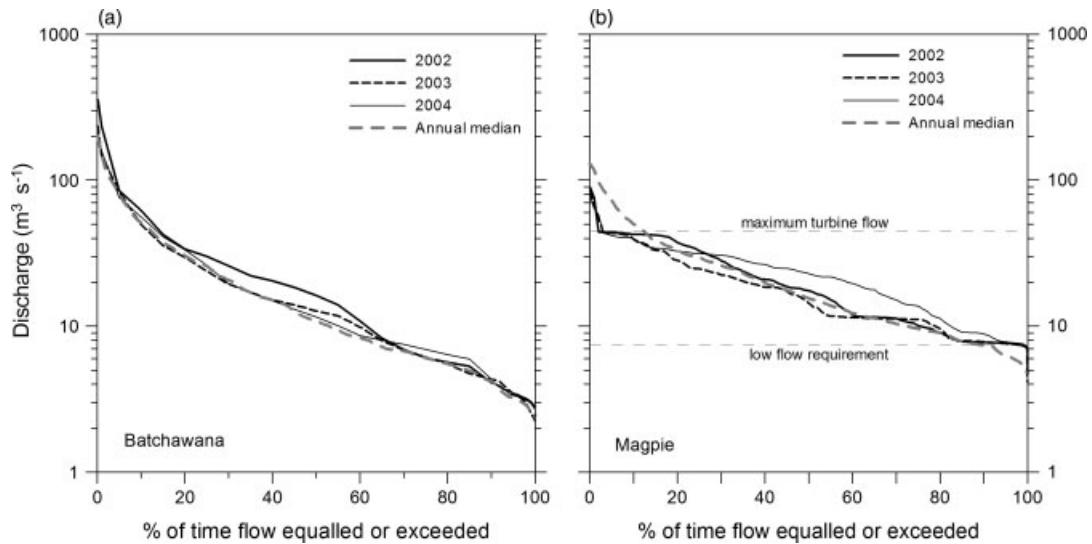


Figure 3. Water year (October to September) flow duration curves (FDCs) for 2002 through 2004 recorded at (a) streamgauge 02BF001 on the Batchawana and (b) Steephill Falls WPF. Also shown for reference is the median of the historical mean annual flow hydrographs for streamgauge 02BF001 on the Batchawana and at streamgauge 02 DB003 on the Magpie.

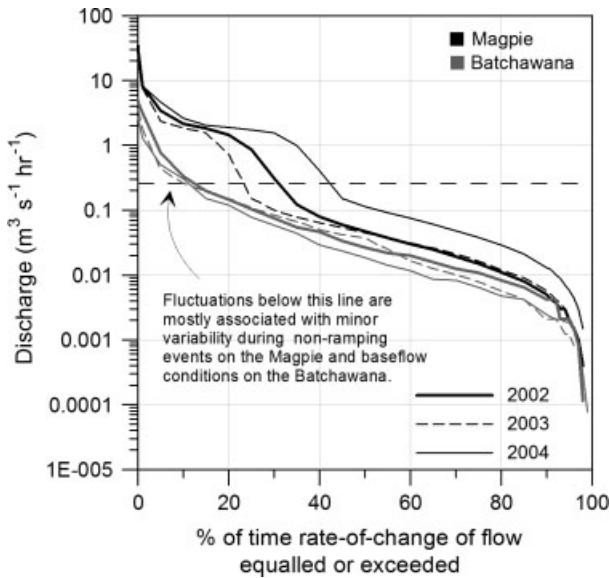


Figure 4. Water year rate of change of flow duration curves using 60-min instantaneous discharge for Steephill Falls WPF and streamgauge 02BF001 on the Batchawana.

three attenuation pools between Steephill Falls WPF and MR-T5 (gauge measuring effective ramping rate) on the rate of change of flow varied depending on the antecedent storage volume in the pools and the volume of water in the peaking event. Although maximum ramping rates were greater at Steephill WPF compared to MR-T5, transitional flows at the latter were of longer duration, owing to the intervening storage capacity of the pools. Individual peaking events reach MR-T5 (2.5 km from WPF) within an hour and take an additional 60 min to reach MR-T15 (7.5 km from WPF) and 90 min to reach MR-T27 (13.5 km from WPF). After the initial flow attenuation before MR-T5, the peaking pattern is transmitted effectively downstream.

Fluvial geomorphology and channel change potential of the Magpie

Under the altered discharge regime, flows at cross-section MR-T27 (13.5 km downstream of Steephill Falls G.S.) typically fluctuate between $10 \text{ m}^3 \text{ s}^{-1}$ (flow stage = 0.59 m) and $45 \text{ m}^3 \text{ s}^{-1}$ (flow stage = 1.33 m). Under these flows, channel width varies between approximately 34.5 and 43.0 m. The flow stage needed to mobilize the D_{16} sized material is 1.81 m ($\sim 80.4 \text{ m}^3 \text{ s}^{-1}$), thus the typical altered discharge range will not initiate bed sediment transport at this cross-section. From 1966 to 1989, flows at MR-T27 exceeded the bed particle mobility threshold ($D_{16} = 80.4 \text{ m}^3 \text{ s}^{-1}$) 5.9% of the time. During the same time period, flows that could transport the median grain sized material (D_{50}) occurred 2.8% of the time. For illustration purposes, only 3 years are presented in Figure 5a (1982, 1983, and 1984). Spring high flows and fall freshets dominate the hydrograph, and occasional summer rains also produce minor to moderate flood peaks during the year. D_{16} sized material is mobilized during every spring freshet and occasionally during the fall freshet. D_{50} sized material is only mobilized during spring freshets and occasionally not at all over the course of a year (e.g. 1984).

Since flow alteration began in 1989, discharges have been greatly controlled, affecting the sediment transport regime of the Magpie downstream of Steephill Falls. Figure 5b displays an entire year of hourly flow data from 2004, overlaid with the bed material transport thresholds. The manipulation of flows has restrained the bedload transport potential at this site. Using hourly flow data, there were only 17 h during 2004 that the discharges exceeded $80.4 \text{ m}^3 \text{ s}^{-1}$ (0.2% of the time). The D_{50} erosion threshold was not exceeded at all during 2004. In contrast, bedload movement in the Batchawana River would be expected during the spring and fall freshets and potentially during long and heavy precipitation events

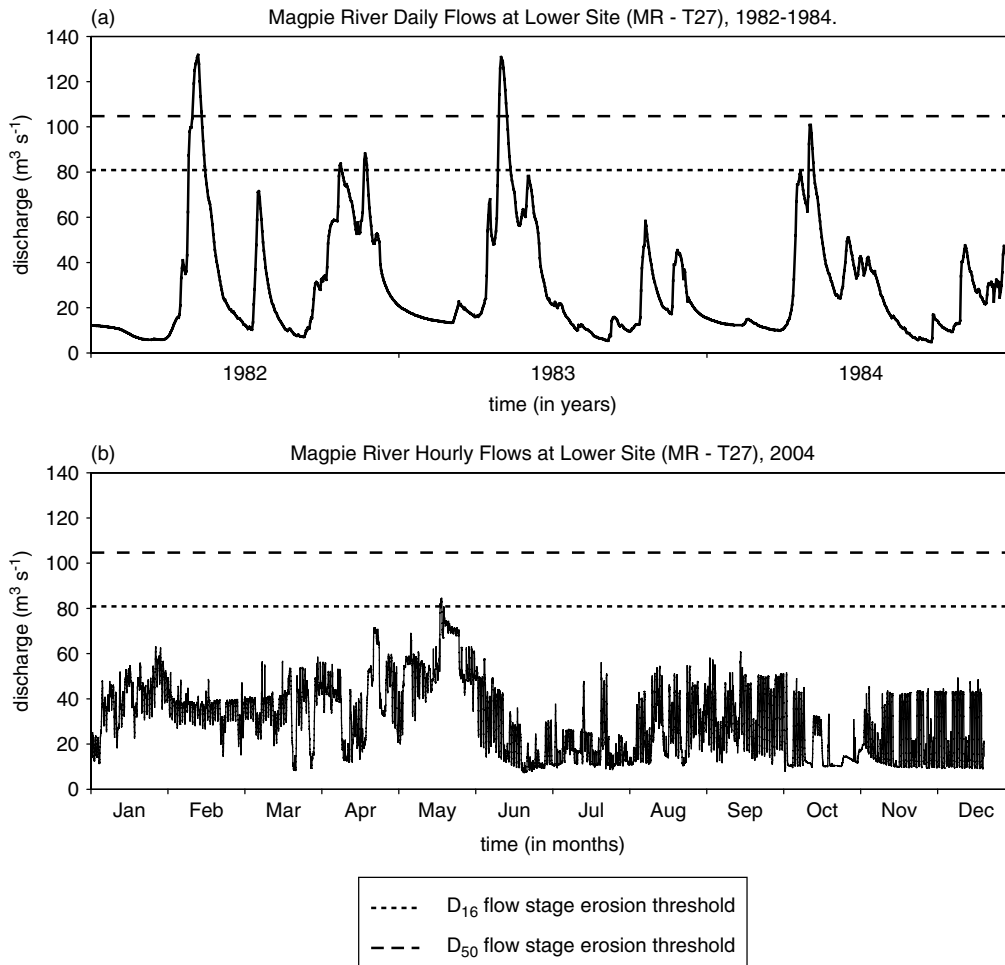


Figure 5. Natural and altered Magpie River flows at the lower site (T27), with bedload entrainment thresholds.

during the summer months, similar to the Magpie River pre-development (Figure 5a).

Benthic invertebrates

Benthic invertebrate community composition, as sampled using surbers in cross-section in the Magpie, was different than that found in the natural Batchawana River (Figure 6). Gastropods and Turbellaria were $\sim 3 \times$ more abundant in the altered Magpie than in the Batchawana whereas Odonata and Plecoptera are rare in the Magpie (Mann–Whitney rank-sum test, gastropods $P \leq 0.001$, Turbellaria $P \leq 0.001$, Odonata $P \leq 0.001$, Plecoptera $P \leq 0.001$).

There are strong lateral gradients in depth, velocity, and frequency of wetting which may influence the abundance and distribution of benthic invertebrates in the varial zone. Substrate sizes were dominated by large gravels (32–64 mm) and were not significantly different among sites and rivers. Diptera, Trichoptera, Ephemeroptera, and Coleoptera increase in abundance towards the deeper, and higher velocity, thalweg in the Magpie while the opposite pattern exists in the Batchawana (Figure 7). In contrast, Oligochaeta, Nematoda, Turbellaria, and Gastropoda decrease in abundance towards the thalweg, whereas, in the Batchawana this decreasing pattern is absent or much reduced (Figure 7).

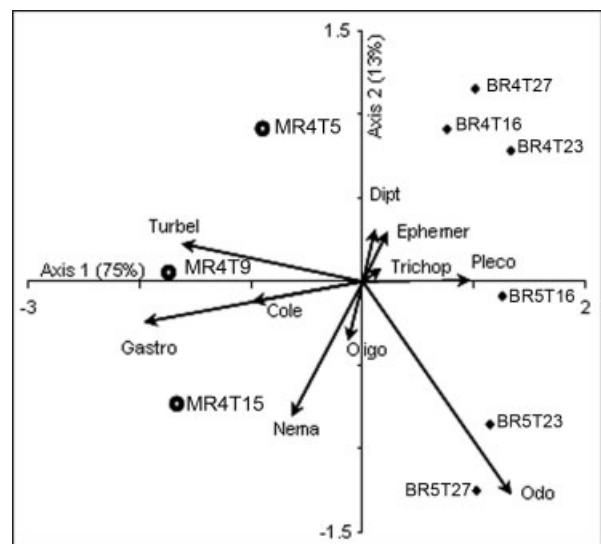


Figure 6. Correspondence analysis (CA) on abundance of the major taxa found in the Magpie and Batchawana Rivers illustrating the similarities and differences in community composition. Magpie River (MAG) at three cross sections (T5, T9, and T15) in 2004 (MAG4T5, MA9G4T, and MAG4T15), and the Batchawana River (B4) sampled in 2004 and (B5) sampled in 2005 at three cross sections (T16, T23, and T27). Diptera (Dipt), Trichoptera (Trichop), Ephemeroptera (Ephermer), Coleoptera (Cole), Oligochaeta (Oligo), Nematoda (Nema), Turbellaria (Turbel), and Gastropoda (Gastro), Plecoptera (Pleco), and Odonata (Odo).

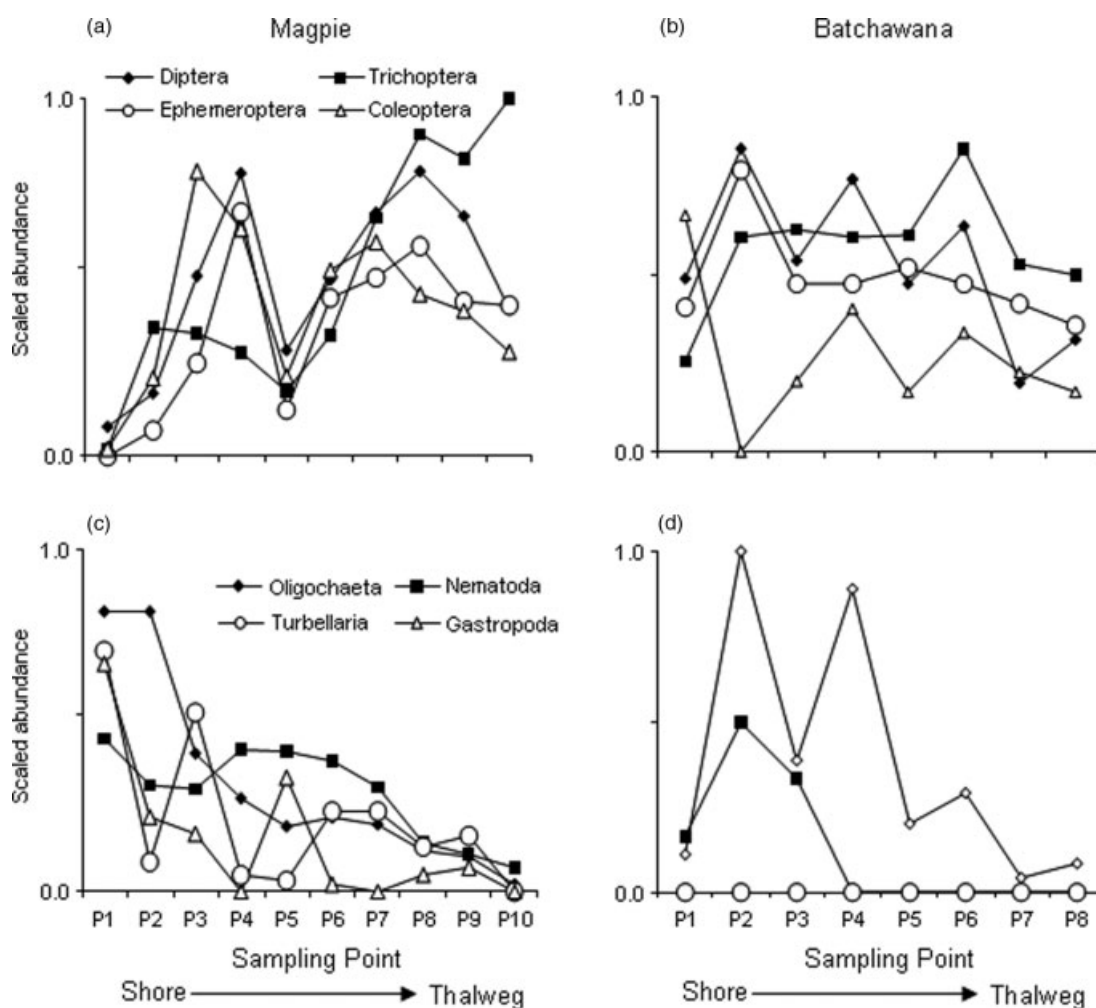


Figure 7. Lateral trends in abundance of benthic invertebrates scaled to 1.0 in relation to samples collected along cross sections in the Magpie (A and C) and Batchawana (B and D) rivers. Top graph shows average pattern for Diptera, Trichoptera, Ephemeroptera, and Coleoptera, and bottom graph shows average pattern for Oligochaeta, Nematoda, Turbellaria, and Gastropoda.

Unlike invertebrate habitat close to shore, the permanently wetted zone of the Magpie had a significantly greater abundance of invertebrates than the Batchawana (river, $F_{1,130} = 87.2$, $P < 0.0001$), although both rivers were on an increasing trend through 2004 (year, $F_{2,130} = 9.7$, $P < 0.001$; Figure 8a). Richness was similar between rivers with both having between 14 and 18 taxa sampled per year, but the Magpie had a significantly greater invertebrate diversity (river, $F_{1,130} = 17.7$, $P < 0.001$) and %EPT (river \times year, $F_{1,130} = 4.6$, $P = 0.01$) than the Batchawana, regardless of the year (Figure 8b–d). This means that across the 3 years, on the Magpie there was between 7 and 18% greater chance that two individuals sampled at random from the population were from different taxa (Figure 8c) and between 8 and 29% more EPT on the Magpie than the Batchawana (Figure 8d). The invertebrate community composition, as indicated by the CA of abundance data from our artificial substrates in the permanently wetted zone, also indicated that the Batchawana had a greater abundance of Odonata, Ephemeroptera, and Plecoptera whereas the Magpie had a greater abundance of Grastropoda and Turbellaria as was found from the invertebrate cross-section sampling.

Fish

Within the Magpie, only VS ($F_{2,180} = 15.1$, $P < 0.001$) and habitat ($F_{1,180} = 9.5$, $P = 0.002$) significantly influenced fish biomass, with VS4 having a lower fish biomass than VS1 or VS3, and fast water habitats having greater biomass than slow water habitats (Figure 9). Within the Batchawana only year influenced biomass, with significantly less biomass being found in 2002 than in 2004 ($F_{2,200} = 6.1$, $P = 0.003$). On average, fish diversity was greater in the Batchawana (PIE = 0.64 ± 0.02 SE) than in the Magpie (PIE = 0.55 ± 0.03 SE) and greater in the slow habitat (PIE = 0.65 ± 0.03) than in the fast habitat (PIE = 0.55 ± 0.02) (ANOVA, river \times year \times habitat: river $F_{1,80} = 9$, $P = 0.004$; habitat $F_{1,80} = 7.6$, $P = 0.007$). Length–weight relationships differed for all species tested (Figure 10). For brook trout, fish were heavier at a given length on the Magpie than they were on the Batchawana (ANCOVA, intercepts $F_{1,110} = 98$, $P < 0.001$); for longnose dace and slimy sculpin, smaller individuals were larger on the Magpie, but on the Batchawana individuals gained weight at a faster rate than the Magpie and the opposite was true for troutperch (ANCOVA

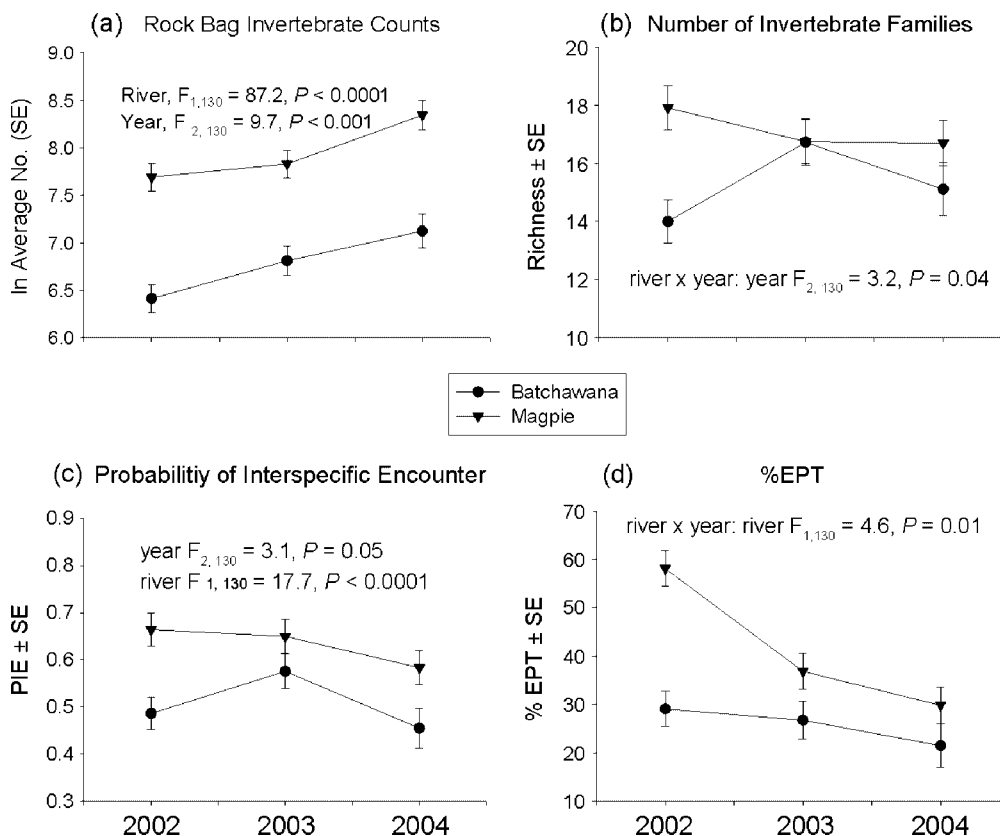


Figure 8. Average (a) abundance (log + 1 transformed), (b) richness, (c) diversity (Probability of Interspecific Encounter—PIE) and (d) %EPT (Ephemeroptera—Plecoptera—Trichoptera) of invertebrates per rock bag \pm standard error (SE) plotted for the Magpie (circles) and Batchawana (triangles) Rivers, 2002–2004.

slopes: LND $F_{1,196} = 48, P < 0.001$, TRP $F_{1,137} = 23$, SLS $F_{1,233} = 8, P = 0.005$; Figure 10).

Stable isotopes

Summary results of the isotope statistical analysis are presented in Table I and Figure 11. Overall, carbon signatures between organisms remained homogenous within natural (mean: -27.4%) and altered rivers (mean: -29.9%) (Figure 11). Most of the variance in $\delta^{13}\text{C}$ was related to differences between taxa (aquatic vegetation: 38%; invertebrates: 53%, fish: 48%) and altered/unaltered sites (invertebrates: 18%; fish: 25%). On average, $\delta^{13}\text{C}$ values were lighter in altered sites compared to unaltered sites (Figure 11) and this difference was particularly marked for invertebrates (2.9%) and fish (3.1%).

Variations in mean $\delta^{15}\text{N}$ signatures were mostly related to trophic fractionation and to differences between altered and unaltered sites (Figure 11). $\delta^{15}\text{N}$ values were significantly higher in altered sites compared to unaltered sites, with the difference being greater at higher trophic levels [from 1.6% (plants) to 2.9% (fish)] (Figure 11). Similar variance partitioning to $\delta^{13}\text{C}$ was observed for the $\delta^{15}\text{N}$ of plants (taxa: 38%), invertebrates (taxa: 35%; reg/unreg.: 33%), and fish (reg/unreg.: 70%; taxa: 13%).

DISCUSSION

Under the constrained dam operations, while it is clear that the hydrograph of the Magpie was significantly

altered relative to a natural hydrograph for the region and that natural bedload transport potential was reduced, many biotic variables remained unaffected when compared to an unaltered system. The combination of reduced ramping rate and minimum discharge appeared to conserve invertebrate abundance and diversity, fish biomass, fish condition, and food web length. Relative to unaltered systems, the abundance and distributions of some sensitive invertebrate taxa were affected. Also affected were fish diversity, the energy base of the food web (carbon signatures), and the nitrogen signatures of many taxa.

The altered streamflow pattern observed at Steephill Falls reflects a cumulative response to variable energy demand and water supply in addition to technological and regulatory constraints. It typifies flow alteration commonly observed downstream of dams and waterpower facilities, in that flow variability is dampened by storing water associated with high flow events for release when water availability is less. The pattern is also dominated by the higher frequency but lower magnitude events typical of waterpower peaking operations. The frequency of peaking events and proportion of time that flows are at the maximum turbine flow or minimum flow requirement ($\sim 25\text{--}40\%$) largely depends on water availability and ramping rate restrictions. With unlimited water supply, ramping restrictions constrain the slope of the middle proportion of the FDC, reflected in the proportion of time at transitional flows.

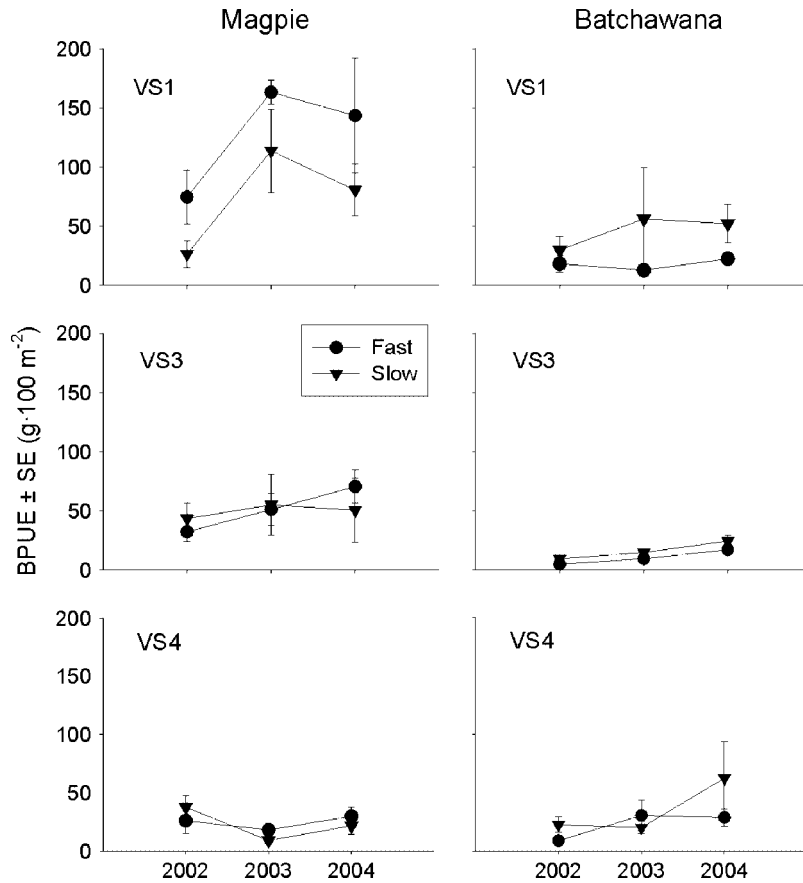


Figure 9. Average fish biomass-per-unit-effort (BPUE in $g\ 100\ m^{-2}$) \pm standard error in valley segments (VS) 1, 3, and 4 for the Magpie and Batchawana Rivers, 2002–2004 showing fast (circles) and slow (triangles) habitat.

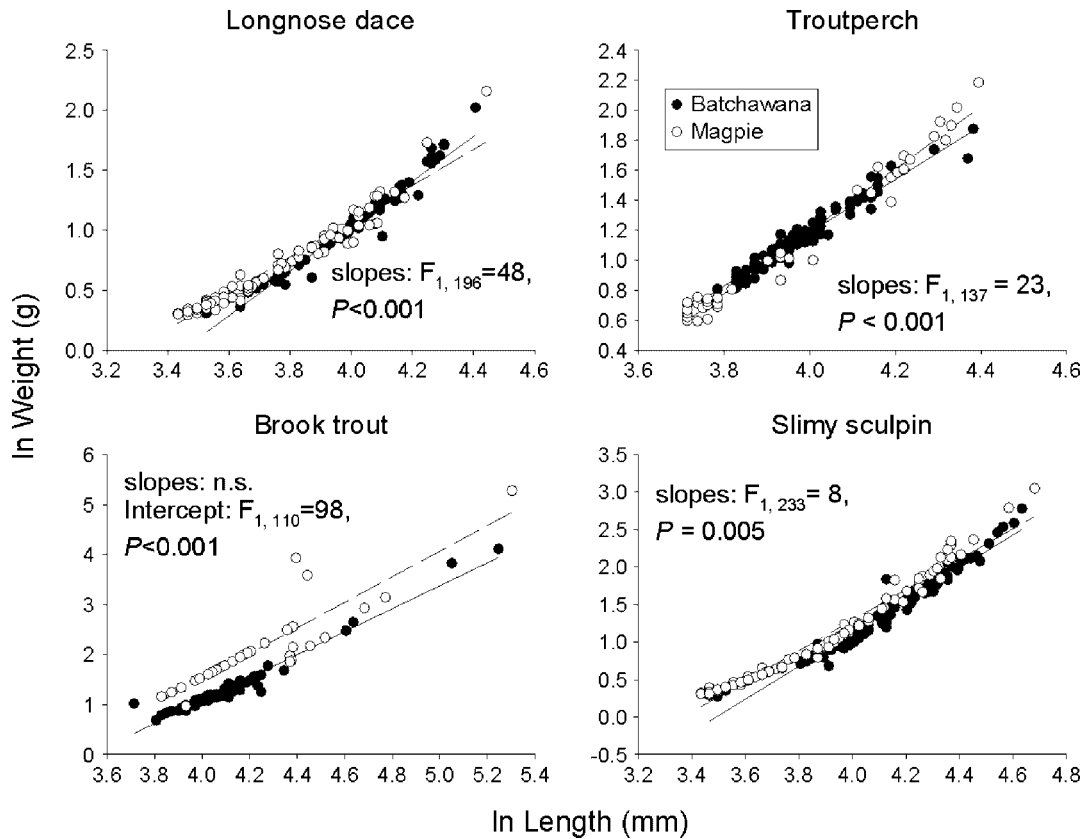


Figure 10. Length (mm)–weight (g) regressions and ANCOVA results comparing relationships between the Magpie (open circles) and Batchawana (closed circles) Rivers for longnose dace, troutperch, brook trout, and slimy sculpin.

Table I. Nested ANOVA of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ for aquatic vegetation, invertebrates, and fish.

		Variable	df	Var. Comp.	F statistic	P value	Per cent variance
Aquatic vegetation	$\delta^{13}\text{C}$	Year	1	0	0.1	0.769	<0.1
		Season (year)	2	0	0.1	0.865	<0.1
		Reg/Unreg (season, year)	2	0	6.4	0.003	<0.1
		Taxa (reg/unreg, season, year)	5	1.8	8.1	<0.001	38.1
		Residual	—	3.0	—	—	61.9
	$\delta^{15}\text{N}$	Year	1	0.4	60.7	<0.001	5.2
		Season (year)	2	3.8	33.1	<0.001	46.0
		Reg/Unreg (season, year)	2	1.2	34.7	<0.001	14.4
		Taxa (reg/unreg, season, year)	5	2.0	16.4	<0.001	23.8
		Residual	—	0.9	—	—	10.7
Invertebrates	$\delta^{13}\text{C}$	Year	1	<0.1	4.4	0.036	0.0
		Season (year)	2	<0.1	15.7	<0.001	0.0
		Reg/unreg (season, year)	4	1.8	19.6	<0.001	17.9
		Taxa(reg/unreg, season, year)	82	5.4	7.9	<0.001	53.3
		Residual	—	2.9	—	—	28.8
	$\delta^{15}\text{N}$	Year	1	<0.1	0.3	0.602	0.0
		Season (year)	2	<0.1	0.0	0.955	0.0
		Reg/unreg (season, year)	4	1.8	44.8	<0.001	33.1
		Taxa (reg/unreg, season, year)	82	1.9	6.0	<0.001	35.5
		Residual	—	1.7	—	—	31.4
Fish	$\delta^{13}\text{C}$	Year	1	0.0	6.5	0.011	<0.1
		Season (year)	2	0.0	2.1	0.121	<0.1
		Reg/unreg (season, year)	4	2.3	49.7	<0.001	25.4
		Taxa (reg/unreg, season, year)	71	4.4	10.9	<0.001	48.3
		Residual	—	2.4	—	—	26.4
	$\delta^{15}\text{N}$	Year	1	0.0	4.2	0.041	<0.1
		Season (year)	2	0.0	22.8	<0.001	<0.1
		Reg/unreg (season, year)	4	2.3	182.1	<0.001	69.8
		Taxa (reg/unreg, season, year)	71	0.4	6.3	<0.001	13.4
		Residual	—	0.5	—	—	16.7

Variance components are estimated using maximum likelihood methods to correct for unbalanced data set.

In many rivers, the pattern of entrainment, transport, and deposition of the bedload component of the total sediment load largely determines channel morphology. From an applied perspective, patterns of bedload transport are critical to maintaining ecological diversity and habitat in stream channels (Martin, 2003). Channel maintenance flows maintain pools, riffles, meanders, and other physical habitats necessary to sustain aquatic ecosystems (Schmidt and Potyondy, 2004). Significant bedload transport in gravel-bed rivers typically begins at discharges approaching bankfull flow (Carling, 1995), and numerous authors have linked the initiation of bedload transport in gravel-bed rivers to some percentage of bankfull flow (Haschenburger and Wilcock, 2003; Ryan *et al.*, 2005).

The Magpie is mostly gravel-bed and cobble-bed with numerous channel bars and vegetated mid-channel bars and occasional steeper riffle sections. The Batchawana is a more incised and entrenched gravel- and boulder-bedded system with many depositional bars and riffle sections, interspersed with deeper pools and bedrock-controlled steeper reaches. Under natural flows at MR-T27, bedload transport would occur during spring and fall freshet flows and during occasional summer discharges related to rainfall events, similar to the Batchawana. The operation of the Steephill Falls G.S. constrained the downstream flows to a range of 10–45 m³ s⁻¹, which

is greatly below the bed particle mobility threshold (80.4 m³ s⁻¹) at the MR-T27 site, thus regulated flows rarely approach a channel maintenance condition. For the same study period (2002–2004), Batchawana River flows annually exceeded the sediment transport threshold for channel maintenance. Some reaches upstream of MR-T27 contain finer grain bed particles and steeper slope; these sites would have greater bed mobility for the same discharges that cause no bedload movement at MR-T27. Patchy surface stone movement is considered common on most rivers during minor flood events, leaving stable stones as refugia for invertebrates even in a river with a high and unstable sediment supply (Matthaei *et al.*, 1999). Sediment in the Magpie has reduced movement relative to an unaltered system. Disturbance theorists propose that disturbance is critical to the structure and dynamics of stream communities (Reice *et al.*, 1990), thus under this theory, a reduction in overall sediment movement would lead to lower benthic diversity.

Our results indicate that Gastropods and Turbellaria were more common in the Magpie than in natural streams, whereas, Odonata and Plecoptera were rare. The rarity of Odonata and Plecoptera from the Magpie may be related to their crawling habit and metamorphoses that take place out of the water (Hynes, 1976; Merritt and Cummins, 1996; Corbet, 1999). These taxa have a

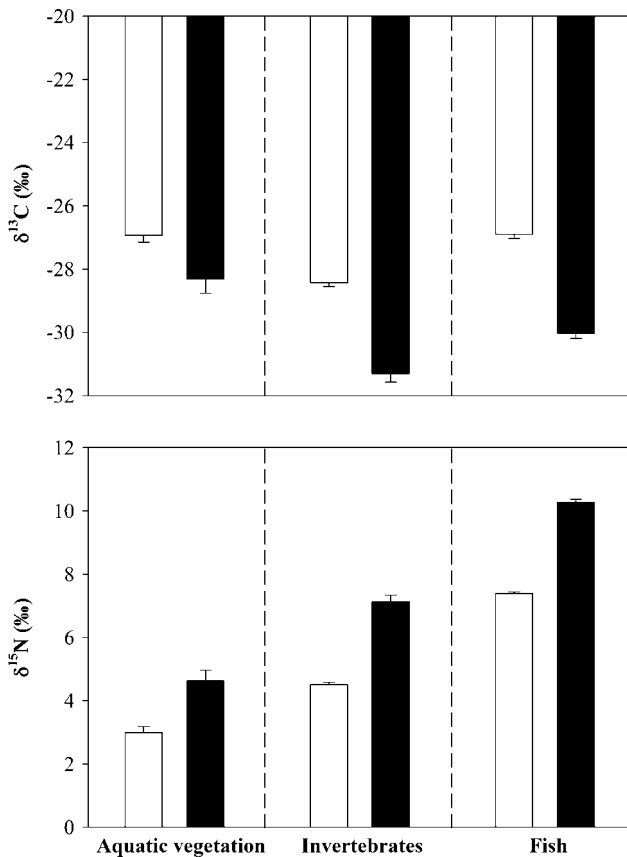


Figure 11. Mean (\pm SE) $\delta^{13}\text{C}$ (top) and $\delta^{15}\text{N}$ (bottom) (‰) for aquatic vegetation, invertebrates, and fish from nonaltered (white) and altered (black) river sites.

limited mobility and may find themselves stranded in the varial zone where predation (e.g. birds) and desiccation of themselves or their eggs could be limiting. Differences were also evident in the distribution and abundance of invertebrates in relation to the distance to the shore, and thus, water depth and velocity, in the varial zone. Areas close to shore in the Magpie contained invertebrates capable of withstanding harsh environmental conditions (e.g. Gastropods, Oligochaeta, and Turbellaria). Snails can retreat into their shells using their operculum which in many species is a kind of trapdoor to close the shell during periods of environmental stress (Thorp and Covich, 1991). Worm and worm-like organisms can burrow into the substrate during low water conditions. More environmentally sensitive invertebrates (e.g. Diptera, Trichoptera, Ephemeroptera, and Coleoptera) were scarce in nearshore areas but common in deeper offshore samples of the Magpie. In contrast, the density of benthos in unaltered natural reference rivers was typically highest near the shoreline. In natural streams, shoreline areas tend to gather more particulate organic matter than adjacent faster thalweg areas where densities of invertebrates are lower. Disturbance frequency, in terms of shear stress and bed movement, might also be higher in the thalweg of these natural streams (Rempel *et al.*, 1999, 2000). In the Magpie, the constant changes in water levels may lead to the washing away and loss of organics in the varial zone. Here, disturbance frequency increases in habitats

shallower and closer to shore, and thus, in the opposite direction found in natural streams.

In this study, we did not sample beyond 10 cm into the hyporheic zone, but it is unlikely that invertebrates in the varial zone of the Magpie burrowed into the riverbed during low flows to escape harsh environmental conditions. In perennial and intermittent streams, del Rosario and Resh (2000) noted that the hyporheic zone is not a refuge when flows cease. Marchant (1988) in his study of the vertical distribution of benthic invertebrates in the bed of the regulated Thomson River Australia found that >72% of invertebrates by number were found in the top 0–10 cm of the riverbed. Although invertebrates are well known to use the hyporheic zone, this phenomenon is not universal in rivers. In the Magpie River, the hyporheic zone may not be well developed because of clogged interstices given the lack of formative floods. Thus, it seems unlikely that invertebrates in the varial zone of the Magpie burrowed into the riverbed during low flows to escape harsh conditions.

Invertebrate standing stock in the thalweg of the Magpie was highly diverse, had a greater %EPT, and was more abundant than that measured in the Batchawana, but it is unclear if results can be attributed to ramping restrictions or minimum flow. This result is contrary to the theory that lack of disturbance in the thalweg on the Magpie would lead to a reduction in invertebrate diversity. Extensive monitoring below nine Tennessee River dams revealed that minimum flows and increases in dissolved oxygen concentrations significantly improved invertebrate family richness and the proportion of intolerant taxa (% EPT), without changes to ramping rate or maximum flow (Bednarek and Hart, 2005). During summer months, the Batchawana flow can fall to extremely low levels (e.g. low of $2.5 \text{ m}^3 \text{ s}^{-1}$ reached in August 2004). Reduced flow can significantly elevate peak summer temperatures (Sinokrot and Gulliver, 2000), decrease wetted area, and allow silt to fill interstitial spaces, thereby reducing habitat heterogeneity for invertebrates (Moog, 1993). Study rock bags were retrieved shortly after summer low flow events (mid-end August), and the associated invertebrate taxonomic composition would be reflective of summer conditions on the rivers. Thus, it is likely that required minimum flow on the Magpie improved invertebrate habitat conditions mid-summer versus the Batchawana. In addition, it is possible that restricted ramping reduced scouring of the substrate, allowing the invertebrate community to proliferate. Parasiewicz *et al.* (1998) found that invertebrate biomass increased by 60% in an alpine Austrian river with the implementation of a minimum base flow and reduced peak flow regulations and attributed the increase to reduced scouring of the substrate during the filling (up-ramping) stage.

The differences in the spatial patterns (longitudinal and lateral) of benthic invertebrates observed in the Batchawana and Magpie point to the importance of where benthic samples are collected. For instance, samples collected in the thalweg will likely be different in composition and abundance than those in the nearshore or shallow

areas of stream. Conclusions drawn from these data without prior knowledge of this distributional pattern would be misleading. For example, only the nearshore areas in the varial zone would be sampled during high flows (i.e. during power generation) because sampling is not possible in the deeper and faster flows for safety reasons. In the Magpie, the invertebrate community in the samples would likely be dominated by Gastropods, Oligochaeta, and Turbellaria, which suggest river impairment. Alternatively, samples could have been collected at low flow, perhaps in the permanently wetted zone, where more sensitive taxa (see above) are common that portrays better health. We suggest that benthic invertebrate sampling in regulated rivers should be as spatially explicit as possible and completed taking into account flow pattern and the definition of the varial and permanently wetted zones.

As a result of proximity to the reservoir, it is not surprising that sites above the dam on the Magpie had the largest fish biomass measures given the dominance of larger bodied species such as burbot (*Lota lota*), yellow perch (*Perca flavescens*) and northern pike (*Esox lucius*) that favour lacustrine habitats (Scott and Crossman, 1973). The fact that sites closer to the dam (on the downstream side) also had significantly greater biomass than sites further downstream, or on the Batchawana, suggests constrained dam operation did not significantly affect the productivity of fish habitats downstream of the dam. Alternatively, habitat productivity may have been enhanced via the supplementation of fish diets by the export of reservoir production as was evidenced by SI signatures on the Magpie. Higher plankton biomass in reservoir has been shown to enhance fish diets close to dams, the influence of which is lessened with distance from the dam (Doi *et al.*, 2008).

Fish diversity was lower on the Magpie despite operational constraints, reflecting the dominance of ubiquitous species (e.g. slimy sculpin and longnose dace). Juveniles of less-common species may have been negatively affected by an overall reduction in the stability, availability, and quality of shallow water habitat on the river margins due to frequent level fluctuations relative to an unaltered river (Bradford, 1997). Preliminary analysis of fish condition via an examination of the length–weight relationships of four common species demonstrated that the brook trout showed the greatest difference between rivers, with fish on the Magpie being heavier at a given length than on the Batchawana. The few studies that have examined the effects of flow alteration on fish growth have found a small but positive effect (e.g. brown trout, Crisp *et al.*, 1983), no effect (e.g. brown trout, Almodovar and Nicola, 1999), or a negative effect hypothesized to be due to a change in suitable habitat (e.g. brown trout, Baran *et al.*, 1995) and/or prey availability (e.g. brown trout, de Crespín de Billy *et al.*, 2002), or simply pulsed flow (e.g. larval suckers in aquaria, Weyers *et al.*, 2003). The apparent positive condition of brook trout on the Magpie may again be due to maintenance of minimum flows that reduce potential heat stress in summer, the abundant and diverse invertebrate food supply in the

permanently wetted zone and/or energy supplementation from the reservoir.

Carbon signatures along the food web remained homogenous within a given river indicating no change in carbon sourcing for invertebrates and fish. Under altered flow regimes, the incorporation of terrestrial detritus and terrestrial prey items is expected to prevail due to the increased connectivity between terrestrial and aquatic environment (Marty *et al.*, 2009). The high portion of $\delta^{13}\text{C}$ variance observed among taxa could reflect variation in the incorporation of terrestrial versus algal carbon. However, terrestrial detritus typically serves as a substrate rather than as a food source for consumers (Finlay, 2001) and consumer's $\delta^{13}\text{C}$ variations are, therefore, more likely related to that of $\delta^{13}\text{C}$ of the aquatic vegetation serving as the food source for lower trophic grazers. Thus, the high variation in invertebrates $\delta^{13}\text{C}$ related to taxonomy may reflect selective feeding on several aquatic vegetation taxa differing in $\delta^{13}\text{C}$ values.

The second most important source of variation in consumer $\delta^{13}\text{C}$ was found between rivers, suggesting potential flow alteration impacts even when ramping restrictions are in place. A previous study reported no effect of flow perturbations on carbon pathways supporting riverine food webs (Marty *et al.*, 2009). Thus, the lighter $\delta^{13}\text{C}$ values observed in the altered river likely results from carbon export from the upstream reservoir where respiration, oxidation, and photolysis are known to deplete carbon signatures to a greater extent than in natural lakes (Marty and Planas, 2008).

As with $\delta^{13}\text{C}$, both taxonomy and river type explained most of the variation in invertebrate and fish $\delta^{15}\text{N}$. Temporal factors explained a significant portion of aquatic plant $\delta^{15}\text{N}$ variation, possibly as a result of changes in nitrogen cycling (Vander Zanden *et al.*, 2005). Nitrogen signatures were significantly higher in the Magpie, likely as a result of higher baseline signatures attributable to microbial mineralization and nitrification/denitrification processes in the upstream reservoir (Vander Zanden and Rasmussen, 2001). Differences may also relate to the temperature variation between sites, responsible for changes in physiological rates (Power *et al.*, 2003; Sweeting *et al.*, 2007). Despite flow constraints in the Magpie, cool water released by the dam from the mid-water layers of the reservoir may influence the riverine food web (Sabater 2008) and explain the differences in $\delta^{15}\text{N}$ between natural and altered sites. The effects of altered thermal regimes on aquatic biota resulting from dam operation include changes in organisms behaviour (Lyon *et al.*, 2008) and physiological rates (Flodmark *et al.*, 2004), that may be tracked by changes in SI signatures (Power *et al.*, 2003), which explain $\delta^{15}\text{N}$ variations between the site types as observed in this study. $\delta^{15}\text{N}$ variations between taxonomic groups reflect both trophic enrichment and differential omnivory (Bearhop *et al.*, 2004). In this study, the trophic position of consumers groups was similar in both river types. The result suggests that flow constraints were able to avoid the reduction in

food web length previously observed under altered flow regimes (Power *et al.*, 1996; Marty *et al.*, 2009).

CONCLUSIONS

Constrained ramping operations result in changes to the flow regime that on the whole are ecologically protective as indicated by some of the measured biological metrics reported here for the Magpie River. Invertebrate abundance and diversity, fish biomass, fish condition, and food web length were all equal to or greater than our unaltered system, and some metrics that differed between rivers, specifically isotope signatures, were attributed to the presence of the reservoir and not to flow alterations. Whether protection is afforded simply by minimum flow restrictions or the combination of minimum flow and ramping rate restrictions remains unclear. The experimental phase of this project (initiated in 2005) removed any ramping restrictions, allowing the WPF operators to ramp as quickly as market demand for electricity requires and water availability allows. The experimental phase of the project is ongoing. Results from both phases of the study will help inform Canadian provincial and federal waterpower guidelines and policy, facilitating science-based decisions regarding ramping at waterpower facilities. In addition, methodologies developed as part of the study will help inform the development of monitoring programs for Water Management Plans at existing and newly developed waterpower facilities in Ontario.

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