RIVER RESEARCH AND APPLICATIONS
River Res. Applic. **33**: 202–213 (2017)
Published online 23 December 2015 in Wiley Online Library
(wileyonlinelibrary.com) DOI: 10.1002/rra.2992

QUANTIFYING THE IMPACT OF WATER ABSTRACTION FOR LOW HEAD 'RUN OF THE RIVER' HYDROPOWER ON LOCALIZED RIVER CHANNEL HYDRAULICS AND BENTHIC MACROINVERTEBRATES

D. ANDERSON[a], H. MOGGRIDGE[a], J. D. SHUCKSMITH[b]* AND P. H. WARREN[c]

[a] *Department of Geography, The University of Sheffield, Sheffield, South Yorkshire, UK*
[b] *Department of Civil and Structural Engineering, The University of Sheffield, Sheffield, South Yorkshire, UK*
[c] *Department of Animal and Plant Sciences, The University of Sheffield, Western Bank, Sheffield, South Yorkshire UK*

ABSTRACT

'Run of the river' (ROR) hydropower schemes have undergone a recent resurgence in Europe, and with legislation requiring the protection and enhancement of the physical and ecological condition of European rivers, there is a need to understand the impacts of these schemes. This paper presents an assessment of the eco-hydraulic impact of a ROR hydropower scheme in the Peak District National Park, UK. Due to the ponded nature of the depleted stretch at the study site, this paper focuses on the characterization of the hydraulic impact of water abstraction for a ROR scheme at the hydropower outlet and samples microhabitats of benthic macroinvertebrates within the hydraulically affected zones. Measurement of hydraulic transects shows that the scheme's operation notably alters river channel hydraulics at 60% of water depth, whilst impacts are much less distinct in close proximity to the river bed. We identify eco-hydraulic relationships between benthic macroinvertebrate communities and localized near-bed velocity and turbulence conditions, thus indicating the potential for water abstraction by ROR schemes to impact lower trophic levels of riverine ecosystems. However, spatial patch-scale (10–100 m^2) meso-habitat comparisons of invertebrate communities around the hydropower outlet showed only subtle differences, suggesting that in this case benthic communities are only minimally impacted by the ROR scheme. © 2015 The Authors. *River Research and Applications* published by John Wiley & Sons, Ltd.

KEY WORDS: hydropower; abstraction; ecohydraulics; invertebrates; run-of-river

Received 16 April 2015; Revised 14 September 2015; Accepted 19 November 2015

INTRODUCTION

Over recent decades, as demand for energy from renewable resources has risen, there has been a dramatic increase in the installation of low head 'run of the river (ROR)' hydropower schemes in both the UK and Europe (Paish, 2002; Environment Agency, 2012). ROR schemes divert a proportion of the river discharge down a secondary channel to a turbine before returning it to the river further downstream. Such schemes are often retrofitted to pre-existing modifications on rivers, such as weirs or historical mill diversions. There is typically no water storage associated with ROR schemes, and it is normally a requirement to maintain a 'hands off' flow, e.g. Q85–95 in the UK, in the natural river channel to maintain ecological function (BHA, 2005), although the scientific justification for this standard is limited. As such, schemes can only operate when there is sufficient discharge in the river to maintain required flows and power the turbine. It is because of such measures that, despite minimal physical evidence, ROR hydropower has been widely accepted as an 'environmentally friendly' form of power production, with the assumption that any impacts on the riverine ecosystem are negligible (Paish, 2002; BHA, 2005). However, recent eco-hydraulic studies have shown ecological impacts from localized changes in velocity and turbulent shear stress (e.g. Gibbins *et al.*, 2010; Blanckaert *et al.*, 2013). Thus, there is clearly a need to understand and quantify the hydraulic and ecological impacts of ROR hydropower. Within Europe, the requirement of the Water Framework Directive (2000/60/EC) for the protection and enhancement of the physical and ecological condition of rivers adds further impetus for this.

The diversion of flow through a ROR scheme creates a stretch of river, extending from the point of abstraction to return, that has depleted flows whilst the scheme is in operation, but a natural flow at other times (hereafter 'depleted stretch'). Thus, scheme operation alters the flow regime and hydraulics of the depleted stretch, and at the point at which the diverted flow re-enters the channel, which has potential consequences for physical habitat (Poff *et al.*, 1997; Biggs *et al.*, 2005), connectivity and, consequently, river ecosystem function and biodiversity (Ward, 1989). The hydraulic and ecological impacts of such schemes are

*Correspondence to: J. D. Shucksmith, Department of Civil and Structural Engineering, The University of Sheffield, Mappin Street Sheffield, South Yorkshire, UK.
E-mail: j.shucksmith@sheffield.ac.uk

commonly assessed through comparison of depleted stretches to 'control' reaches (Kubecka *et al.*, 1997; Copeman, 1997) or predicted through hydrological modelling (Copeman, 1997; Lamouroux *et al.*, 2015). Such studies have shown changes to hydraulic habitat; depleted reaches have reduced proportions of lotic habitat (Kubecka *et al.*, 1997; Ovidio *et al.*, 2008) and lower water velocity in pools and riffles (Anderson *et al.*, 2006), although overall habitat heterogeneity is not compromised (Copeman, 1997). However, despite the importance of hydraulics in shaping riverine ecosystems, studies reporting detailed, spatialized hydraulic measurements are very uncommon (Ovidio *et al.*, 2008, Lamouroux *et al.*, 2015).

Ecological studies on ROR hydropower have largely investigated impacts on fish, e.g. Kubecka *et al.* (1997) and Ovidio *et al.* (2008). However, the hydraulic impact of ROR schemes may have consequences at lower trophic levels, for example affecting benthic macroinvertebrates which have important roles in organic matter processing and are a vital food source for fish, riparian insects, birds and mammals (Castella *et al.*, 1995; Covich *et al.*, 1999). Many invertebrate taxa are known to be sensitive to near-bed hydraulic conditions; velocity and turbulence parameters influence critical processes such as food acquisition, movement (drift) between habitat patches, substratum condition and predator evasion (Hart and Finelli, 1999; Jowett, 2003), whilst also causing entrainment or dislodgement (Hart and Finelli, 1999) and, sometimes, physical damage to taxa (Growns and Davis, 1994). Thus, many invertebrate taxa express streamwise (longitudinal) velocity preferences (Extence *et al.*, 1999) and are known to be vulnerable to hydraulic changes associated with water abstraction, particularly taxa requiring high velocity habitats (Degani *et al.*, 1993), or which have restricted feeding (and thus habitat) requirements, e.g. filter feeding, net spinning Trichoptera (Jowett, 2000). Invertebrate distributions can also be affected by shear stress (Gibbins *et al.*, 2010) and turbulence (Blanckaert *et al.*, 2013), although this is not always the case (Robson *et al.*, 1999), indicating that the response may vary among rivers (Jowett, 2000).

The few existing studies of invertebrate responses to ROR schemes compare invertebrate communities between depleted and control reaches. These show variable responses: Jesus *et al.* (2004) observed a notable reduction of Ephemeroptera, Plecoptera and Trichoptera ('EPT') taxa, Copeman (1997) found a reduction of Ephemeroptera species, whilst Kubecka *et al.* (1997) found no measureable impact. In the absence of baseline data (i.e. enabling before-after comparison), attributing any impacts directly to the scheme is challenging, particularly in heavily modified water bodies where multiple anthropogenic stressors (e.g. physical modification and water pollution) may influence the riverine environment (Anderson *et al.*, 2014).

Improving our understanding of the potential impacts of ROR hydropower will clearly require a range of approaches. One of these, which has not yet been utilized, is to use a fine-scaled, interdisciplinary approach, focused on the detailed *in situ* characterization of both hydraulic modification and biotic pattern, within an individual ROR scheme. This paper takes such an approach, presenting an assessment of the localized eco-hydraulic impact of a ROR scheme in the Peak District National Park, UK, using benthic macroinvertebrates as ecological indicators. Specifically, our objectives are as follows: (1) quantify the local hydraulic impacts of a ROR scheme on the river channel; (2) investigate the eco-hydraulic relationships between benthic macroinvertebrates and hydraulic conditions; and (3) draw conclusions regarding the potential scale of eco-hydraulic impacts of ROR schemes on benthic macroinvertebrates.

MATERIALS AND METHODS

Study site

The village of Alport is located in the south eastern part of the Peak District National Park, Derbyshire, UK (Figure 1). The hydropower scheme, installed in 2008, was retrofitted to a disused mill on the River Lathkill (Qmean $1.17\,m^3\,s^{-1}$, Paish and Needle, 2008). The scheme makes use of a pre-existing natural tufa weir, heightened to 3 m, to divert a maximum discharge of $1.05\,m^3\,s^{-1}$ to a 30 kW Crossflow turbine whilst ensuring that a minimum residual flow of $0.12\,m^3\,s^{-1}$ (Q95) is maintained in the depleted stretch whenever the scheme is operational (Paish and Needle, 2008). Flow duration statistics, which define the scheme operation, are based on data from a flow gauging station 2 km downstream, which has been operational since 1998 (Paish and Needle, 2008).

In this case, the depleted stretch is characterized by a series of small weirs (ranging from 0.5 to 1.0 m), creating a succession of small weir ponds. Although the habitat in these ponds may be affected by the change in discharge, it was considered that the scheme's operation has the most significant hydraulic impact immediately downstream of the hydropower outlet, where water from the hydropower scheme meets the depleted stretch, forming a confluence. The study therefore focused on the hydraulic and ecological characterization of this confluence zone. This paper assesses the impact of the scheme's operation by presenting profiles of temporally averaged velocity and turbulence statistics of the channel at the hydropower outlet when the scheme was running, and not running, recorded under similar river discharge conditions. In addition, benthic invertebrate community composition is examined at several locations within the study area, and the relationship between these samples and

DOI: 10.1002/rra

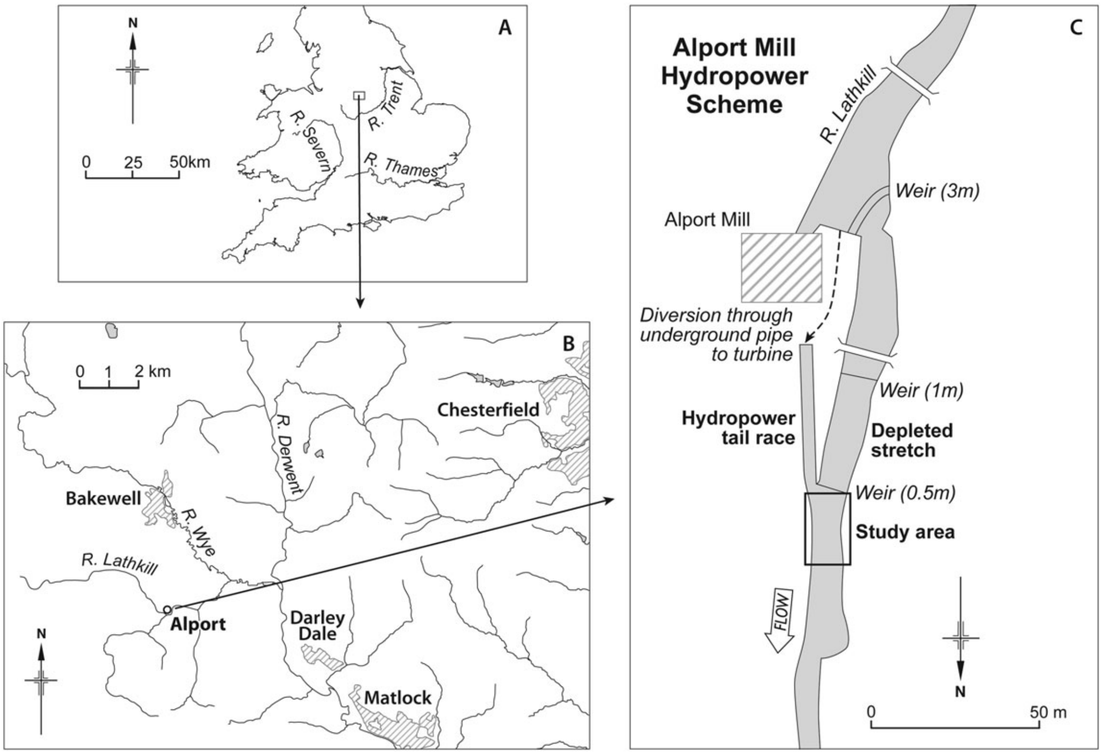

Figure 1. Location maps of (A) the Peak District National Park, (B) the study site, Alport, and (C) a schematic of the hydropower scheme highlighting the study area.

near-bed velocity measurements taken at the sampling locations are examined.

Hydraulic characterization of the hydropower scheme

Hydraulic characterization of the site was conducted via direct velocity measurements obtained using a 3D Nortek Vectrino, downward facing, acoustic Doppler velocimeter (ADV). The ADV probe was attached to an adjustable surveying tripod, allowing the probe to be stably positioned at any point in the river. The probe was capable of taking simultaneous measurements of three orthogonal velocity components at a frequency of 25 Hz at a given point, providing data for temporally averaged velocity as well as standard turbulent statistics. A convergence test was conducted to determine that reliable mean velocity values and a good representation of turbulence could be obtained by measurements at each point. Individual measurements consisted of 100–300 s sampling period, the time being dependent on the hydraulic complexity of the sampling location. Raw ADV data were processed in WinADV 32 (Wahl, 2000), applying the phase space threshold despiking filter (Goring and Nikora, 2002) and minimum value filters for correlation (60) and SNR (15) parameters.

In order to assess the hydraulic impact of the scheme's operation, transverse profiles of velocity were obtained at 60% of water depth, at ~0.5 m intervals along transects located 5 and 15 m downstream of the confluence apex (Figure 2). After processing, these measurements were used to calculate temporally averaged streamwise velocity and Turbulent Kinetic Energy (Eqn 1), used in this case to indicate the overall level of turbulence at each location.

$$TKE = 0.5\left(\overline{u'^2} + \overline{v'^2} + \overline{w'^2}\right) \tag{1}$$

where TKE is turbulent kinetic energy ($m^2 s^{-2}$), u' is the fluctuation from the mean of the longitudinal velocity, v' is the fluctuation from the mean transverse velocity, and w' is the fluctuation from the mean of the vertical velocity ($m s^{-1}$).

It was not possible to control the operation of the hydropower scheme to allow measurements of hydro on/off to be made under identical flow conditions. However, it was possible to obtain a series of comparable measurements under moderate to high-flow conditions in autumn 2012 and spring 2013. Measurements were taken on six different days with a varying proportion of flow passing through the hydropower scheme (Table I). For each 'case' discharge measurements were made using a calibrated ValePort Electromagnetic Current Meter and the 1-point velocity cross-sectional area method detailed in British Standard

DOI: 10.1002/rra

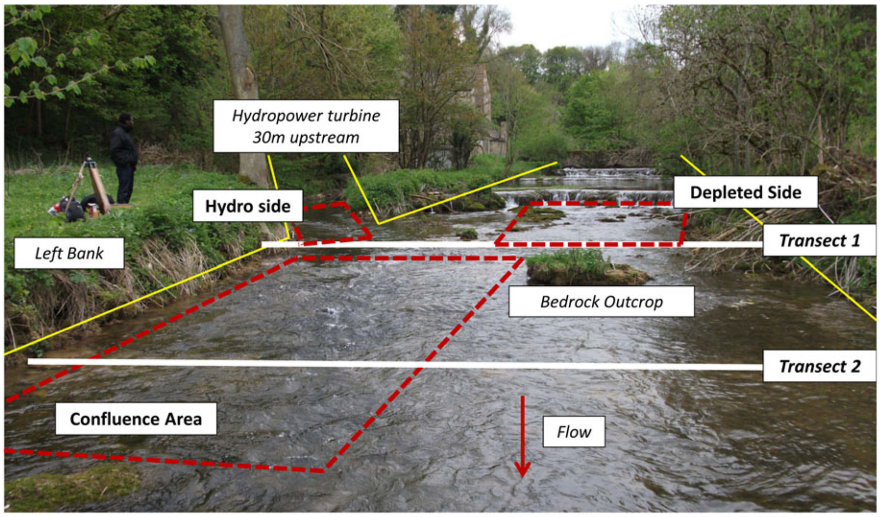

Figure 2. An annotated photograph of the study area at Alport Mill (May 2013). Yellow lines indicate channel banks, red dashed boxes indicate the three discrete invertebrate sampling areas, white lines indicate hydraulic transects. This figure is available in colour online at wileyonlinelibrary.com/journal/rra

BS EN ISO 748:2007 (British Standards, 2007). Because of the relatively high river discharge conditions, the measured impacts are assumed to be near the minimum for the scheme, as hydraulic changes during lower river discharges, when higher proportions of the flow are diverted, are likely to be more notable.

Invertebrate sampling

The study area was split into three sampling areas distinctly affected by the scheme's operation (Figure 2): the hydropower (hydro) side, excavated when the scheme was installed (quadrat 2.7 × 11.5 m); the depleted side, which is depleted when the scheme is in operation (6.5 × 11.5 m); and the 'confluence area', where the majority of the two water bodies meet (4.4 × 11.5 m) (topographical variation in the channel partially separates the flows up and downstream).

Within each sampling area, six randomly located, 30 s 'kick' samples were collected (each sampling approximately 0.25 m^2 of streambed) in May 2012, using a 1 mm mesh net. This was a point when the scheme had been running most of the time over the preceding winter and spring, resulting in sustained exposure of the invertebrate community to any altered hydrological conditions resulting from the operation of the installation. Samples were preserved in 70% industrial methylated spirits prior to sorting and identification to family level (Wallace *et al.*, 1990; Dobson *et al.*, 2012). Taxa were assigned to functional feeding groups (FFG) (Merritt *et al.*, 2002) and family richness; Shannon–Weiner Diversity and %EPT (Ephemeroptera, Plecoptera, and Trichoptera) were calculated for each individual sample. Percent EPT was analysed as these taxa are thought to be particularly sensitive to disturbances in riverine ecosystems (McKay and King, 2006).

Near bed hydraulic characterisation

Near bed velocity and turbulent shear stress have been found to be relevant to benthic macroinvertebrates (Gibbins *et al.*, 2010; Blanckaert *et al.*, 2013). In order to investigate the potential eco-hydraulic impacts, additional near bed ADV

Table I. Discharge and abstraction scenarios for ADV transect sampling flow exceedance data taken from Paish and Needle [24]

Case	Date of Sampling	River discharge ($m^3 s^{-1}$)	Flow exceedance probability	Discharge through scheme ($m^3 s^{-1}$)	Discharge through scheme %
1	18/10/2012	1.56	Q20	0	0
2	05/10/2012	1.09	Q38	0.24	22
3	23/10/2012	1.09	Q38	0.34	31
4	05/04/2013	0.96	Q41	0.59	62
5	19/03/2013	1.06	Q38	0.73	69

DOI: 10.1002/rra

measurements were obtained. At each invertebrate sampling location measurements were taken at 5 and 6 cm above the river bed (≈20% of the total flow depth, following Allen and Castillo (2007) and Gibbins *et al.* (2010)). The velocity and turbulence parameters were depth averaged to provide a representation of hydraulic conditions at each sampling location. It is recognized that macroinvertebrates may experience a wider proportion of the hydraulic boundary layer than can be feasibly sampled; however, measurements were designed to provide a representation of boundary layer conditions suitable for subsequent analysis. Measurements were undertaken on four occasions ('hydrological scenarios'): twice with the scheme off and twice with it on, with 85 and 55% of the discharge diverting through the scheme respectively (Table II). Measurements were taken under moderate flow conditions (Q50–Q77) to capture hydraulic variability predominantly associated with the operation of the scheme and were taken opportunistically over 16 months following the invertebrate samples.

The measurements are used to calculate temporally averaged streamwise velocity (ū), TKE (Eqn 1) and turbulent shear (Reynolds) stresses (Eqns 2 and 3).

$$\tau_{uv} = -\rho\overline{u'v'} \tag{2}$$

$$\tau_{uw} = -\rho\overline{u'w'} \tag{3}$$

where τ_{uv} and τ_{uw} are the horizontal and vertical Reynolds stresses respectively ($\mathrm{N\,m^{-2}}$), ρ is the density of water ($\mathrm{kg\,m^{-3}}$). For subsequent patch scale analysis the hydraulic parameters were spatially averaged within the three sampling areas, to provide an overview of the conditions under each of the four hydrological scenarios. For ordination analysis, averages of the hydraulic measurements over the four hydrological scenarios were taken at each sampling location and combined with the taxonomic data. Although these point measurements cannot quantify the full range of hydraulic conditions experienced by the macro-invertebrates (which may be situated at various points within the boundary layer and be exposed to a wider range of conditions because of temporal fluctuations in flow), the averaging of velocity and turbulence parameters under different quantifiable flow and operational scenarios provides a representation of the prevailing hydraulic conditions at each sampling location such that a ordination analysis can be carried out.

In addition to hydraulic measurements, four water quality parameters were measured at the sampling locations using a YSI Professional Plus multimeter: dissolved oxygen, total dissolved solids, conductivity and pH. These were recorded on four separate occasions, twice with the scheme on and twice with it off. However, all showed negligible differences between sampling locations and so were not included in subsequent analyses.

Statistical analysis

For patch scale comparisons of taxonomic and hydraulic data, normality of data was assessed using the Shapiro-Wilk test. Data following non-normal distributions were 1/x transformed to achieve normality. Statistical differences between sampling areas were tested using one-way ANOVA with Bonferoni corrected post-hoc pairwise comparisons.

Canonical Correspondence Analysis (CCA) was used to investigate eco-hydraulic links between hydraulic parameters and invertebrate community composition. Prior to ordination, taxonomic data were $\log_{10}(x+1)$ transformed to avoid over influence by highly abundant taxa. Rare taxa, i.e. those occurring in only one sample (five taxa), were removed to reduce noise (Bailey *et al.*, 2004) and avoid result biasing by rare species (Cao *et al.*, 2001). The environmental covariates consisted of mean near-bed longitudinal velocity, vertical and horizontal Reynolds stress and Turbulent Kinetic Energy.

All statistical tests were performed in XL Stat 2013.

RESULTS

Hydraulic impact of the scheme

The results from transect 1, 5 m downstream of the confluence apex, indicate that the operation of the hydropower scheme noticeably alters the hydraulic characteristics of the immediate area around the hydropower outlet. However,

Table II. Discharge and abstraction scenarios for ADV invertebrate point sampling Flow exceedance data taken from Paish and Needle [24]

				Discharge through scheme	
Case	Date of sampling	River discharge ($\mathrm{m^3\,s^{-1}}$)	Flow exceedance probability	($\mathrm{m^3\,s^{-1}}$)	%
6	14/03/2013	0.81	Q50	0.45	55
7	03/06/2012	0.52	Q71	0.44	85
8	09/08/2013	0.43	Q74	0	0
9	20/09/2013	0.36	Q77	0	0

results from transect 2, a further 10 m downstream, suggest that the magnitude of the impact is quickly reduced.

Results from transect 1 (Figure 3) suggest that when the hydropower scheme is off, under high discharge conditions (case 1), there is little variability in longitudinal velocity across the channel. This trend changes when the hydropower scheme is operational; the velocity profile becomes distinctly less uniform, with higher velocities on the hydropower ('hydro') side (0–2.7 m from left bank) and lower velocities on the depleted side (3.3–7.0 m from left bank). The extent to which this occurs appears dependent on the degree of abstraction.

When the scheme is operational (cases 2–5), the temporally and spatially averaged velocity ($<\bar{u}>$) measured on the hydro side is $0.20\,\mathrm{m\,s^{-1}}$ higher than when the scheme is off (case 1), despite the off case being under higher river discharge conditions. Whilst the increase in $<\bar{u}>$ between low ($\leq$31% of the total flow, cases 2–3) and high abstraction ($>$60% of total flow diverted, cases 4–5) scenarios was $0.28\,\mathrm{m\,s^{-1}}$ and higher velocities ($>1.0\,\mathrm{m\,s^{-1}}$) were found to extend further across the channel under high abstraction. In contrast, on the depleted side, the reduction in $<\bar{u}>$ between low and high abstraction scenarios was $0.18\,\mathrm{m\,s^{-1}}$. The impact of increased water abstraction had little impact on the velocity between 3.5 and 5 m from left bank, but velocities were notably reduced (up to $0.6\,\mathrm{m\,s^{-1}}$) by high abstraction regimes between 5.5 and 7.0 m from left bank. Such reductions result in a lotic areas being transformed into lentic areas.

Figure 3 also suggests the creation of a turbulent confluence between 1.7 and 3.9 m from left bank when the scheme is operational, as a result of water bodies of different speeds (hydropower tail race and depleted stretch) merging. Peaks in TKE were present in this area for all on scenarios, yet absent for the off scenario. Peak TKE was found to increase with degree of abstraction. TKE is notably variable on the depleted side under all study conditions because of the influence of complex bed topography (i.e. exposed bedrock), so no clear trends could be identified.

Results from transect 2 (Figure 4) suggest that operation of the hydropower scheme still impacts river channel hydraulics 15 m downstream of the confluence apex. However, the magnitude of the impact is reduced. In particular, Figure 4 highlights that the impact on the velocities of the hydro side (0–4.0 m from left bank) is less pronounced, although there is a notable reduction in velocities between 0 and 1.5 m from left bank and the formation of a secondary high velocity area between 3.5 and 4.0 m from left bank when the scheme is operational (cases 2–5). Again, the extent of these hydraulic impacts appears to increase with the proportion of water abstracted by the scheme. Figure 4 further suggests that increased levels of abstraction result in notable reductions in temporally and spatially averaged velocity and TKE on the depleted side (4.4–6.4 m from left bank). Under high abstraction cases, temporally and spatially averaged velocity and TKE are reduced by 0.11 and $1.1\times10^{-4}\,\mathrm{m^2\,s^{-2}}$ respectively, in comparison to low abstraction scenarios.

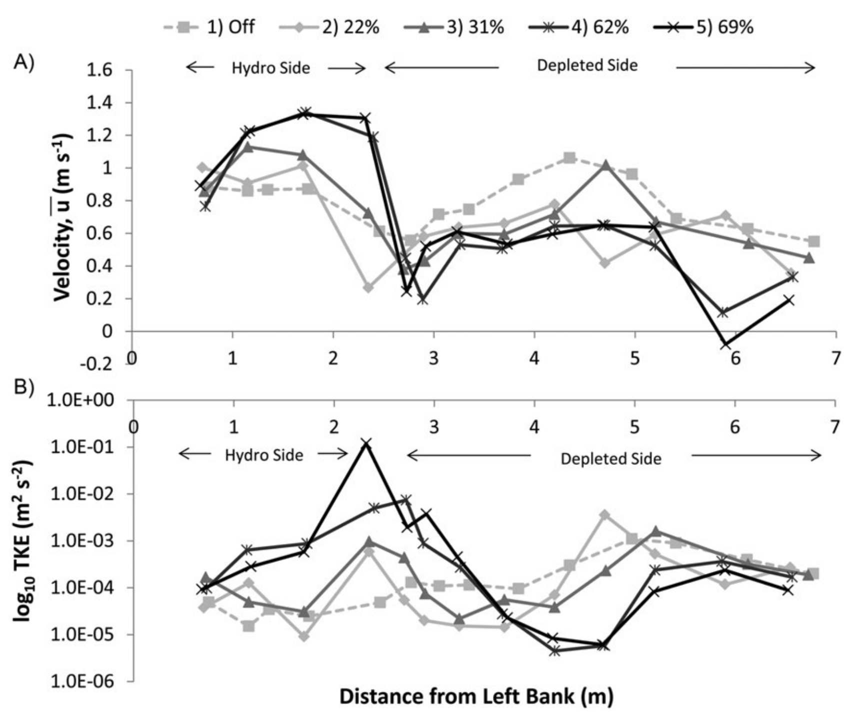

Figure 3. Transverse profiles of temporally averaged: (A) streamwise velocity, (B) TKE, at transect 1 under a variety of abstraction conditions. The legend shows % discharge diverting through the scheme on each occasion and numbering corresponds to the case numbers in Table I.

DOI: 10.1002/rra

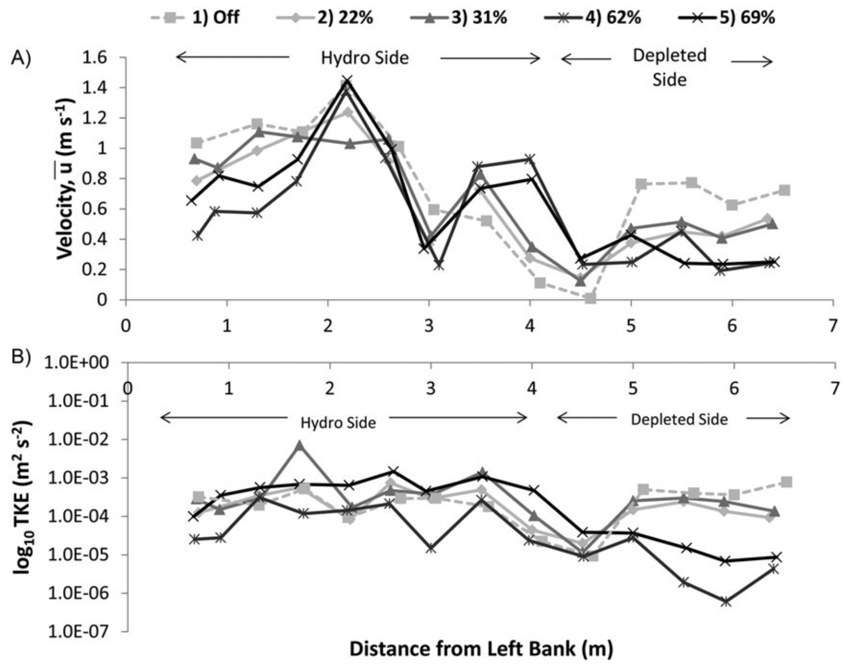

Figure 4. Transverse profiles of temporally averaged: (A) streamwise velocity, (B) TKE, at transect 2 under a variety of abstraction conditions. The legend shows % discharge diverting through the scheme on each occasion and numbering corresponds to the case numbers in Table I.

Near bed hydraulics and patch scale macro invertebrates

Near-bed ADV point measurements at the invertebrate sampling locations (Figure 5) indicate that the average near-bed velocity and vertical Reynolds stress within the sample areas decreases from confluence area to hydro side to depleted side when the scheme is operational. However, there is considerable variability within sampling areas for both parameters. When the scheme is off, ADV measurements indicate that the confluence area and depleted side are generally more hydraulically comparable, although considerable variation remains within each sampling area. In contrast, there is a noticeable reduction in both the variability and magnitude of both parameters in the hydro side sampling area when the scheme is off resulting in the area becoming noticeably more lentic. Further to this, Figure 5 shows that under moderate depletion (case 7), the average velocity and vertical Reynolds stress experienced within the depleted sampling area are comparable to the other sampling quadrats. However, the depleted sampling area average is notably reduced, in comparison to the other areas, for both parameters under high depletion (case 6). This is significant for Reynolds stress (one-way ANOVA, $F=4.19$, d.f.$_t=2.15$, $p=0.036$) but only marginally significant for velocity (one-way ANOVA,

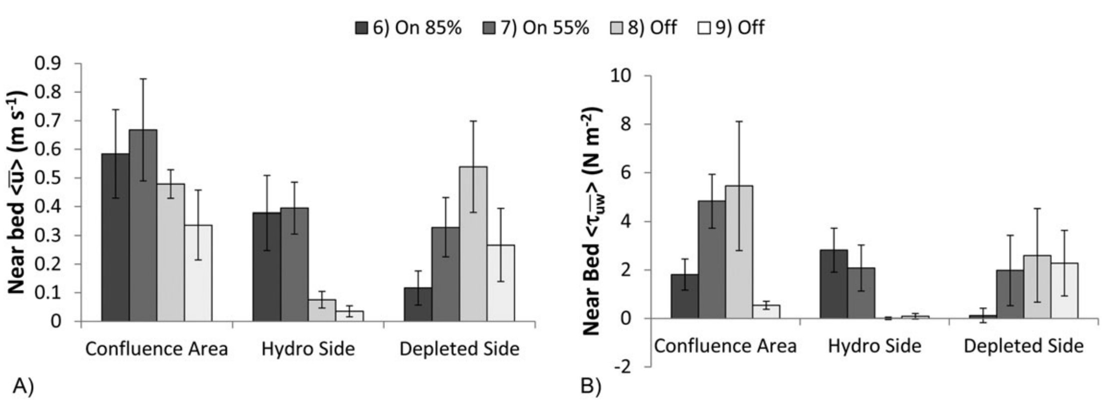

Figure 5. Bar plots showing the mean (±1 SE): (A) near-bed velocity and B) vertical shear stress of the grouped invertebrate sampling points on each of the four sampling days. The legend shows % discharge diverting through the scheme on each occasion and numbering corresponds to the case numbers in Table II.

DOI: 10.1002/rra

$F = 3.67$, d.f.$_t = 2.15$, $p = 0.05$), and there is still distinct within sample area variability under high abstraction conditions.

Patch scale comparisons of the taxonomic data (Figure 6) show average family richness ($F = 2.16$, d.f. = 2.15, $p = 0.15$), Shannon–Weiner diversity ($F = 1.38$, d.f. = 2.15, $p = 0.28$) and %EPT taxa ($F = 1.45$, d.f. = 2.15, $p = 0.27$), do not differ significantly between sample areas (one-way ANOVA). Functionality of the patch scale communities also appears largely unaffected with little difference in the proportion of the various FFG. There is a suggestion of a reduction in the average proportion of the FFG collector–filterers (Merritt *et al.*, 2002) in the depleted sampling area, although this is not statistically significant (one-way ANOVA, $F = 2.53$, d.f. = 2.15, $p = 0.11$). This is attributable to marked reductions in Simuliidae abundance. In the hydro side and confluence area quadrats, Simuliidae contribute ~11 and 12% of total abundance respectively, whilst in the depleted sampling area this is reduced to 1.5%. However, when Simuliidae are excluded, the depleted sampling area supports the highest average proportion of collector-filterers.

Eco-hydraulic relationships between macroinvertebrates and near-bed hydraulics

The CCA analysis of the individual invertebrate samples and hydraulic parameters (mean velocity, Reynolds stresses and TKE averaged over the hydrological scenarios) is shown in Figure 7. The first two axes of the CCA explained 67.98% of the total variance in the data set. Axis 1 accounted for 36.60% of the variance and was strongly associated with increasing TKE (regression coefficient: 1.370), decreasing vertical Reynolds stress (−0.924) and near-bed velocity (−0.518). The second axis accounted for 31.38% of the variance and was strongly associated with magnitude horizontal Reynolds stress (−1.271). These correlations are consistent with expectations for hydraulic measurements at a site featuring a confluence, where large-scale turbulent structures and lateral momentum transfer are driven by the horizontal influx of water from the secondary channel travelling at a different velocity to the main stream (Rhoads and Sukhodolov, 2008).

The separation of the samples based on the hydraulic parameters (Figure 7) shows some distinction between zones along the second axis because of variation in horizontal shear stress. The confluence samples showed the highest negative values, indicative of high horizontal shear stress towards the left bank, which would be expected from the tailrace joining the channel from the right bank. However, there is high variability within the sampling zones. This variability is also evident along the first axis, which shows a strong negative gradient between TKE and velocity and vertical Reynolds stress. This gradient has the greatest influence on invertebrate community composition and eco-hydraulic relationships are evident. In particular, the trichopteran families Limnephilidae, Rhyacophilidae and Philopotamidae appeared to be strongly associated with samples from high velocity habitats, as did the filter-feeding dipteran larvae Simuliidae. In contrast, collector-gatherer families such as Asellidae, Caenidae and Ephemeridae were associated with samples from low velocity and higher TKE habitats, whilst

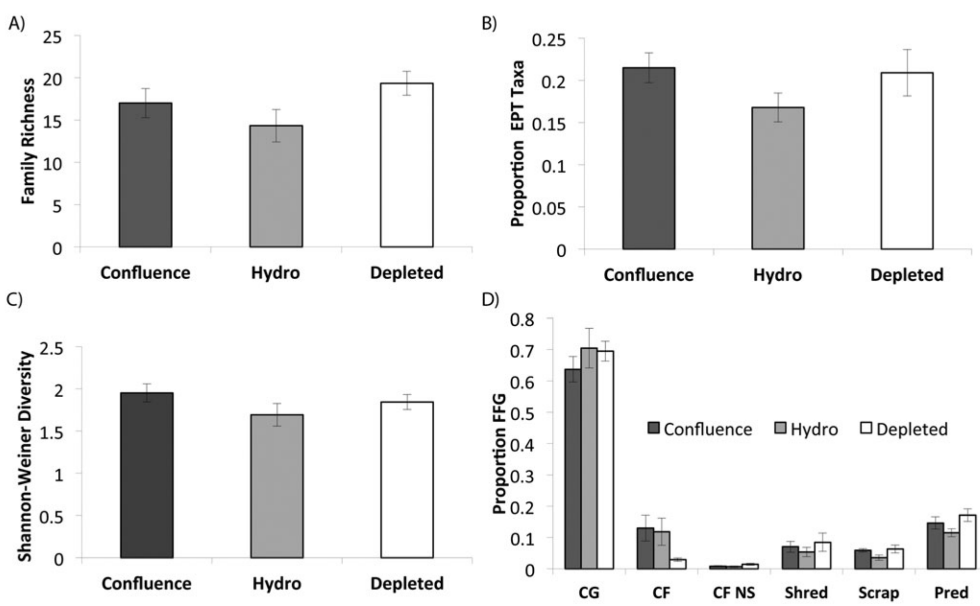

Figure 6. Bar plots showing the mean (of the six individual samples, ±1 SE): (A) family richness, (B) proportion EPT taxa, (C) Shannon–Weiner diversity, (D) proportion contributed by each functional feeding group (FFG) for each sampling quadrat, where CG = collector–gatherers, CF = collector–filterers, CF NS = collector–filterers excluding Simuliidae, Shred = shredders, Scrap = scrapers, Pred = predators.

DOI: 10.1002/rra

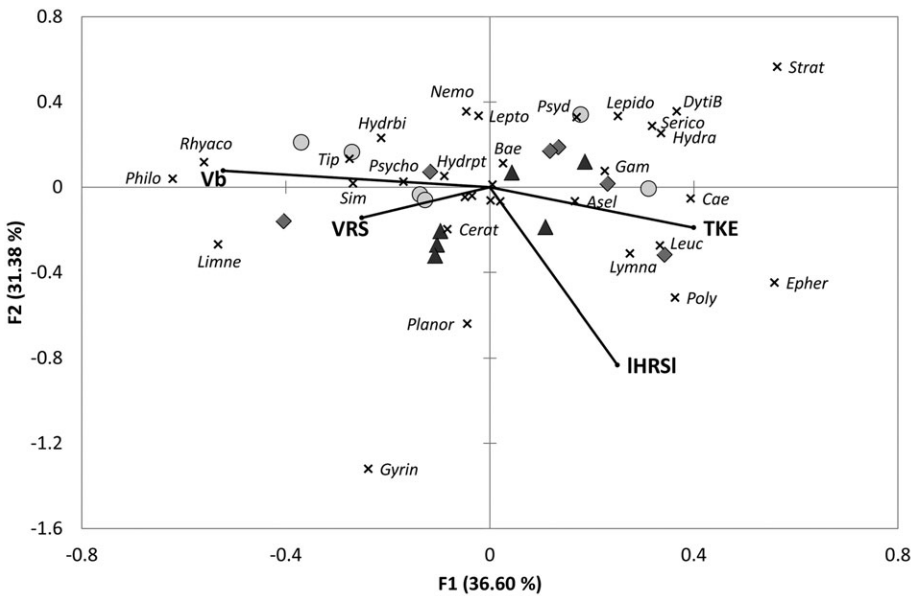

Figure 7. A triplot of the first two canonical axes from a CCA of the invertebrate community data explaining 67.98% of the variance within the data set. Sample sites are represented by filled shapes; triangles (hydro), diamonds (depleted) and circles (confluence). Quantitative hydraulic covariates are indicated by lines and invertebrate taxa by crosses with associated identification code (see appendix). Vb is near-bed velocity ($m\,s^{-1}$), VRS is the vertical shear stress ($N\,m^{-2}$), $|HRS|$ is the magnitude horizontal shear stress ($N\,m^{-2}$) and TKE is turbulent kinetic energy ($m^2\,s^{-2}$).

families including Stratiomyidae and Dytiscidae, were absent or in low abundance in samples from high velocity and vertical and horizontal Reynolds stress areas. Most of the samples from the depleted side of the channel experienced lower velocity and vertical Reynolds stress and supported higher numbers of taxa that are more tolerant of low flow velocity habitats (e.g. Asellidae and Polycentropodidae). In contrast, most samples from the confluence area experienced higher velocity and Reynolds stresses and supported taxa tolerant of these conditions, e.g. Rhyacophilidae and Philopotamidae. Samples from the hydropower side supported increased proportions of taxa with less specific flow preferences, which may be a consequence of the area alternating between a fairly lentic environment when the scheme is off and strongly lotic one when it is on.

DISCUSSION

The results of the hydraulic study showed that operation of the scheme in moderate-high river flow conditions caused noticeable alterations to hydraulic conditions at 60% water depth around the hydropower outlet. Five metres downstream of the confluence apex, increasing levels of abstraction resulted in more lentic conditions on the depleted side of the channel, with notable decreases in streamwise velocity, whilst the hydro side exhibited marked increases. Hydraulic impacts from the operation of the scheme were also detected a further 10 m downstream, although these were less pronounced on the hydro side because of the natural shift of the thalweg towards the left bank because of a bedrock outcrop (Figure 2).

The subsequent eco-hydraulic investigation showed that invertebrate community composition was related to variations in near-bed streamwise velocity, TKE and vertical Reynolds stress. There was an ecological separation between sites with high TKE and high horizontal Reynolds stress and those with high velocity and vertical Reynolds stress, likely because of the unusual hydraulic nature of the field site (i.e. the presence of a confluence). Thus, macroinvertebrates within high TKE sites are likely to be influenced by the effects associated with turbulence, rather than velocity, such as fluxes of fine particulate organic matter and dissolved gases (Bouckaert and Davis, 1998). The high proportion of collector-gatherer and shredder feeders found in the high TKE sites, which feed primarily on FPOM, supports this. These findings complement other studies, which have shown the importance of turbulence in influencing benthic invertebrate composition (Bouckaert and Davis, 1998; Gibbins *et al.*, 2010; Blanckaert *et al.*, 2013).

Whilst there was high variability between samples, impacts from the hydropower scheme were evident. Two thirds of samples from the depleted area were associated with low velocity and vertical Reynolds stress. This may result from operation of the hydropower scheme, because of increased periods of reduced flow, causing decreased near-bed velocity and Reynolds stress, and favouring fine sediment and organic matter deposition. Invertebrate composition of these

DOI: 10.1002/rra

samples showed high proportions of taxa, which prefer more lentic environments, such as collector-gatherer and shredder feeders (Extence *et al.*, 1999; McKay and King, 2006). These findings support Degani *et al.*'s (1993) suggestion that it is the high-velocity favouring taxa which suffer most as a result of water abstraction.

Despite the identified eco-hydraulic linkages and the notable hydraulic impacts associated with increasing water abstraction, results from the invertebrate samples suggest only minor differences between the communities of the three discrete sample areas when studied at the family level, despite the depleted side quadrat experiencing considerable depletion when the scheme operates. No significant differences were observed in family richness, Shannon–Weiner diversity or %EPT taxa between sample areas. The only notable community differences were a reduction in abundance of Simuliidae and increased abundance of Psychodidae and Stratiomyidae in the depleted side sample area, though as Simuliidae represent 86.3% of all filter-feeding taxa in the study area, this may suggest a significant change in community function. Whilst not significant, the higher proportion of collector–gatherer and shredder feeders associated with many of the depleted side samples also indicates that hydropower operation could impact community function.

There is, of course, some possibility that lack of clear differences may be in part because of the family level resolution of the data, if there are in fact differences in species composition within families between the sample sites. However, flow preferences are still evident at the family level (Dolédec *et al.*, 2007), and our observations during sample processing suggest that most families we recorded are not composed of many recognizably different taxa or that those are distributed differently between sample areas.

Whilst these findings suggest that the depleted side community differences could result from depletion from the hydropower scheme, we cannot entirely separate this effect from other possible influences: the sample areas, whilst in close proximity, may have supported different communities prior to the installation of the scheme as a result of other environmental variation. For instance, the depleted side quadrat is predominantly tufa bedrock, whilst substrate in the other sampling areas is a mixture of tufa, gravels and pebbles. The lack of more pronounced patch scale effects could also be attributed to the large near-bed hydraulic variability within the sample areas. Whilst the operation of the hydropower scheme had marked hydraulic impacts mid-way up the water column, such impacts were much less uniform near the bed, where hydraulics are of known importance to benthic invertebrates (Hart and Finelli, 1999; Jowett, 2003). Under moderate depletion (55% reduction in depleted stretch), the three sample areas remained comparable in terms of mean near bed velocity and vertical Reynolds stress. Under high abstraction (85% reduction), the average of both parameters was diminished in the depleted quadrat, but there was still variability in both parameters within all sample areas. This suggests that depletion may not have any greater impact on near-bed hydraulics (and thus invertebrates) than other local influences, such as bed topography, substrate size and aquatic vegetation (Biggs *et al.*, 2005; Jowett, 2003; Allan and Castillo, 2007).

In the case of this scheme, any impacts on channel hydraulics and the invertebrate community are likely to be localized to our study area, as existing weirs within the depleted stretch create a continuously ponded habitat under all conditions, in which invertebrates tolerant of low velocity and turbulence are likely to dominate. Had the depleted stretch not been impounded, the operation of the hydropower scheme may have considerably reduced the amount of habitat available within the stretch for invertebrates favouring high-velocity and Reynolds stress conditions. Thus, impacts from such schemes may differ in less modified, free-flowing systems.

In summary, our study highlights that the distinct impact water abstraction for ROR schemes can have on mid-water-column hydraulics of depleted areas. As our results only capture the impact under moderate-high river flow conditions and below-capacity abstraction regimes, the observed hydraulic changes are likely to represent the minimum impacts of the scheme. The study provides further evidence to support the importance of near-bed hydraulic conditions in helping structure benthic invertebrate communities, but highlights that such conditions may be less affected by water abstraction for ROR schemes than higher in the water column. Whilst we detect a relationship between invertebrates and near-bed hydraulic conditions around the hydropower installation, community differences between sample areas were not significant. The depleted area supported increased proportions of lentic taxa relative to other sampling areas, but differences in community composition were only subtle, supporting the findings of Copeman (1997) and Kubecka *et al.* (1997). The marked reductions of EPT taxa noted by Jesus *et al.* (2004) were not observed, although their study scheme had been in operation for over 10 years, in contrast to 3 years in our study system, meaning community response to the impacts may be incomplete (Petts *et al.*, 1994).

The observed hydraulic alterations caused by operation of the ROR scheme do highlight the potential for such schemes to affect fundamental riverine processes. In combination with evidence of eco-hydraulic interactions between benthic invertebrates and near-bed hydraulic conditions, such impacts may have consequences for riverine biota, although these effects may be localized and moderated by other natural and anthropogenic habitat influences. These impacts should be considered in the design and assessment of ROR hydropower schemes, although further research to isolate and assess the relative importance of these impacts is required.

ACKNOWLEDGEMENTS

This research was supported by an Engineering and Physical Sciences Research Council 'Energy-Futures' research studentship. The authors would also like to thank the Peak District National Park for additional funding and support, Haddon Estate for site access, The University of Bradford School of Engineering Design and Technology for ADV loan, Soha Alsolmy for invertebrate checking, David McCutcheon and Shaun Oliver for assistance in figure design and the many people who assisted in the field.

REFERENCES

Allan D, Castillo M. 2007. *Stream Ecology: Structure and Function of Running Waters*. Springer: Dordrecht, The Netherlands.

Anderson E, Freeman M, Pringle C. 2006. Ecological consequences of hydropower development in Central America: impacts of small dams and water diversion on neotropical stream fish assemblages. *River Research and Applications* **22**: 397–411.

Anderson D, Moggridge H, Warren P, Shucksmith J. 2014. The impacts of 'run-of-river' hydropower on the physical and ecological condition of rivers. *Water and Environment Journal* **29**: 268–276.

Bailey R, Norris R, Reynoldson T. 2004. *Bioassessment of Freshwater Ecosystems: Using The Reference Condition Approach*. Springer-Verlag: New York.

BHA. 2005. *A Guide to UK Mini-Hydro Developments*. The British Hydropower Association. Available at: http://www.british-hydro.org/ (Accessed 12 September 2015).

Biggs B, Nikora V, Snelder T. 2005. Linking scales of flow variability to lotic ecosystem structure and function. *River Research and Applications* **21**: 283–298.

Blanckaert K, Garcia X, Ricardo A, Chen Q, Pusch M. 2013. The role of turbulence in the hydraulic environment of benthic invertebrates. *Ecohydrology* **6**: 700–712.

Bouckaert F, Davis J. 1998. Microflow regimes and the distribution of macroinvertebrates around stream boulders. *Freshwater Biology* **40**: 77–86.

British Standards. 2007. BS EN ISO 748 Hydrometry – Measurement of liquid flow in open channels using current-meters or floats.

Cao Y, Larsen D, Thorne R. 2001. Rare species in multivariate analysis for bioassessment: some consideration. *Journal of the North American Benthological Society* **20**: 144–153.

Castella E, Bickerton M, Armitage P, Petts G. 1995. The effects of water abstractions on invertebrate communities in UK streams. *Hydrobiologia* **308**: 167–182.

Copeman V. 1997. The impact of micro-hydropower on the aquatic environment. *Water and Environment Journal* **11**(6): 431–435.

Covich A, Palmer M, Crowl T. 1999. The role of benthic invertebrate species in fresh water ecosystems. *Bioscience* **49**(2): 119–127.

Degani G, Herbst G, Ortal R, Bromley H, Levanon D, Netzer Y, Harari N, Glazman H. 1993. Relationship between current velocity depth and the invertebrate community in a stable river system. *Hydrobiologia* **263**: 163–172.

Dobson M, Pawley S, Fletcher M, Powell A. 2012. *Guide to Freshwater Invertebrates*. Freshwater Biological Association: Ambleside.

Dolédec S, Lamouroux N, Fuchs U, Mérigoux S. 2007. Modelling the hydraulic preferences of benthic macroinvertebrates in small European streams. *Freshwater Biology* **52**: 145–164. DOI:10.1111/j1365-2427200601663x.

Environment Agency. 2012. *Hydropower Good Practice Guidelines*. Environment Agency: Bristol.

Extence C, Balbi D, Chadd R. 1999. River flow indexing using British benthic macroinvertebrates: a framework for setting hydroecological objectives. *River Research and Applications* **15**(6): 543–574.

Gibbins C, Batalla R, Vericat D. 2010. Invertebrate drift and benthic exhaustion during disturbance: response of mayflies (Ephemeroptera) to increasing shear stress and river-bed instability. *River Research and Applications* **26**: 499–511.

Goring D, Nikora V. 2002. Despiking acoustic doppler velocimeter data. *Journal of Hydraulic Engineering* **128**(1): 117–126.

Growns I, Davis J. 1994. Longitudinal changes in near-bed flows and macroinvertebrate communities in a western Australian stream. *Journal of the North American Benthological Society* **13**: 417–438.

Hart D, Finelli C. 1999. Physical–biological coupling in streams – the pervasive effects of flow on benthic organisms. *Annual Review of Ecology and Systematics* **30**: 363–395.

Jesus T, Formigo N, Santos P, Tavares G. 2004. Impact evaluation of the Vila Viçosa small hydroelectric power plant (Portugal) on the water quality and on the dynamics of the benthic macroinvertebrate communities of the Ardena River. *Limnetica* **23**: 241–256.

Jowett I. 2000. Flow management. In *New Zealand Stream Invertebrates: Ecology and Implications for Management*, Collier K, Winterbourne M (eds). The New Zealand Limnological Society: Christchurch; 289–312.

Jowett I. 2003. Hydraulic constraints on habitat suitability for benthic invertebrates in gravel-bed rivers. *River Research and Applications* **19**: 495–507.

Kubecka J, Matena J, Hartvich P. 1997. Adverse ecological effects of small hydropower stations in Czech Republic: 1 Bypass plants. *Regulated Rivers: Research & Management* **13**: 101–113.

Lamouroux N, Gore J, Lepori F, Statzner B. 2015. The ecological restoration of large rivers needs science-based predictive tools meeting public expectations: an overview of the Rhône project. *Freshwater Biology* **60**: 1069–1084. DOI:10.1111/fwb12553.

McKay S, King A. 2006. Potential ecological effects of water extraction in small unregulated streams. *River Research and Applications* **22**: 1023–1037.

Merritt R, Cummins K, Berg M, Novak J, Higgins M, Wessell K, Lessard J. 2002. Development and application of a macroinvertebrate functional-group approach in the bioassessment of remnant river oxbows in southwest Florida. *Journal of the North American Benthological Society* **21**(2): 290–310.

Ovidio M, Capra H, Philippart J. 2008. Regulated discharge produces substantial demographic changes on four typical fish species of a small salmonid stream. *Hydrobiologia* **609**: 59–70.

Paish O. 2002. Small hydro power: technology and current status. *Renewable and Sustainable Energy Reviews* **6**: 537–556.

Paish O, Needle J. 2008. *Alport Mill Mini-hydro Scheme: Environmental Statement*. Derwent Hydro: Duffield, Derbyshire.

Petts G, Maddock I, Bickerton M, Ferguson A. 1994. Linking hydrology and ecology: the scientific basis for river management. In *The Ecological Basis for River Management*, Harper DM, Ferguson AJD (eds). Wiley: Chichester.

Poff NL, Allan D, Bain M, Karr J, Prestegaard K, Richter B, Sparks R, Stromberg J. 1997. The natural flow regime. *Bioscience* **47**(11): 769–784.

Rhoads BL, Sukhodolov AN. 2008. Lateral momentum flux and the spatial evolution of flow within a confluence mixing interface. *Water Resources Research* **44**: W08440. DOI:10.1029/2007WR006634.

Robson B, Chester E, Davis J. 1999. Manipulating the intensity of near-bed turbulence in rivers: effects on benthic invertebrates. *Freshwater Biology* **42**: 645–653.

Wahl T. 2000. Analyzing ADV data using WinADV. *Joint Conference on Water Resources Engineering and Water Resources Planning and Management American Society of Civil Engineers,* July 30-August 2 2000, Minneapolis, Minnesota.

Wallace I, Wallace B, Philipson G. 1990. *A Key to the Case-Bearing Caddis Larvae of Britain and Ireland*. Freshwater Biological Association: Ambleside.

Ward J. 1989. The four dimensional nature of lotic ecosystems. *Journal of the North American Benthological Society* **8**(1): 2–8.

APPENDIX
CCA CODES FOR INVERTEBRATE TAXA.

Invertebrate Code	Macroinvertebrate Family	Invertebrate Code	Macroinvertebrate Family
Asel	Asellidae	Lepido	Lepidostomatidae
Bae	Baetidae	Lepto	Leptoceridae
Cae	Caenidae	Leuc	Leuctridae
Cerat	Ceratopogonidae	Limne	Limnephilidae
Chiro	Chironomidae	Lymna	Lymnaeidae
DytiB	Dytiscidae	Nemo	Nemouridae
Elm	Elmidae	Philo	Philopotamidae
Emp	Empididae	Planor	Planorbidae
Emphem	Ephemeridae	Poly	Polycentropodidae
Epheml	Ephemerellidae	Psycho	Psychomyiidae
Gam	Gammaridae	Psyd	Psychodidae
Gyrin	Gyrinidae	Rhyaco	Rhyacophilidae
Hydra	Hydrachnidia	Serico	Sericostomatidae
Hydrbi	Hydrobiidae	Sim	Simuliidae
Hydro	Hydropsychidae	Strat	Stratiomyidae
Hydrpt	Hydroptilidae	Tip	Limoniidae

DOI: 10.1002/rra