

See discussions, stats, and author profiles for this publication at: <http://www.researchgate.net/publication/269727208>

Ecological impacts of small hydropower plants on headwater stream fish: from individual to community effects

ARTICLE *in* ECOLOGY OF FRESHWATER FISH · DECEMBER 2014

Impact Factor: 1.59 · DOI: 10.1111/eff.12210

DOWNLOADS

89

VIEWS

220

6 AUTHORS, INCLUDING:



Lluís Benejam

University of Vic

37 PUBLICATIONS 381 CITATIONS

SEE PROFILE



Carolina Solà

Generalitat de Catalunya

24 PUBLICATIONS 299 CITATIONS

SEE PROFILE



Antoni Munné

Catalan Water Agency. (Generalitat de Cata...

58 PUBLICATIONS 673 CITATIONS

SEE PROFILE



Emili García-Berthou

Universitat de Girona

121 PUBLICATIONS 3,326 CITATIONS

SEE PROFILE

Ecological impacts of small hydropower plants on headwater stream fish: from individual to community effects

Lluís Benejam^{1,2,3}, Sandra Saura-Mas^{4,5,2}, Mònica Bardina⁶, Carolina Solà⁶, Antoni Munné⁶, Emili García-Berthou³

¹BETA Technology Centre, Aquatic Ecology Group, University of Vic – Central University of Catalonia, Vic, Spain

²Center for Study of Terrestrial and Aquatic Conservation Biology (CEBCAT-La Balca), 17600 Figueres, Catalonia, Spain

³Institute of Aquatic Ecology, University of Girona, 17071, Girona, Catalonia, Spain

⁴CREAF (Center for Ecological Research and Forestry Applications), 08193 Cerdanyola del Vallès, Catalonia, Spain

⁵Unit of Ecology, Department of Animal and Plant Biology and Ecology, Autonomous University of Barcelona, 08193 Bellaterra, Catalonia, Spain

⁶Catalan Water Agency (ACA), 08036 Barcelona, Catalonia, Spain

Accepted for publication November 15, 2014

Abstract – Hydroelectricity is increasingly used worldwide as a source of renewable energy, and many mountain ranges have dozens or hundreds of hydropower plants, with many more being under construction or planned. Although the ecological impacts of large dams are relatively well known, the effects of small hydropower plants and their weirs have been much less investigated. We studied the effects of water diversion of small hydropower plants on fish assemblages in the upper Ter river basin (Catalonia, NE Spain), which has headwater reaches with good water quality and no large dams but many of such plants. We studied fish populations and habitat features on control and impacted reaches for water diversion of 16 hydropower plants. In the impacted reaches, there was a significantly lower presence of refuges for fish, poorer habitat quality, more pools and less riffles and macrophytes, and shallower water levels. We also observed higher fish abundance, larger mean fish size and better fish condition in the control than in impacted reaches, although the results were species-specific. Accordingly, species composition was also affected, with lower relative abundance of brown trout (*Salmo trutta*) and Pyrenean minnow (*Phoxinus phoxinus*) in the impacted reaches and higher presence of stone loach (*Barbatula barbatula*) and Mediterranean barbel (*Barbus meridionalis*). Our study highlights the effects of water diversion of small hydropower plants from the individual to the population and community levels but probably underestimates them, urging for further assessment and mitigation of these ecological impacts.

Key words: water diversion; weir; fish condition; brown trout *Salmo trutta*; Pyrenean streams; Ter river basin

Introduction

Freshwater ecosystems are among the most altered ecosystems worldwide by multiple, interacting pressures including pollution, water abstraction and the construction of weirs and dams (Pires et al. 1999; Xenopoulos & Lodge 2006; Ayllón et al. 2012). The main purpose of weirs and dams includes irrigation for agriculture, water provisioning for human use and hydropower generation. Although most energy consumed worldwide currently is from fossil fuels and nuclear power plants, 10.5% comes from so-called

renewable sources, of which hydropower is the most important (Habit et al. 2007; EIA 2011; Koç 2012). For instance, in Spain, about 8.1% of the total energy produced by 2008 came from hydroelectricity, and there are over 1300 hydropower plants (Espejo & García 2010). Increasing energy consumption in the next decades, combined with the need to reduce hydrocarbon-based energy production, has renewed the interest in hydropower generation (Koç 2012) and an increase in the number of hydropower plants that are being built or planned (Nilsson et al. 2005; Zhang et al. 2012). Nonetheless, with the potential to

Correspondence: L. Benejam, BETA Technology Center, Aquatic Ecology Group, University of Vic, Sagrada Família 7, 08500 Vic, Catalonia, Spain.
E-mail: lluis.benejam@uvic.cat

doi: 10.1111/eff.12210

1

accommodate further large projects becoming limited, many countries are increasingly focusing on the development of small hydropower plants (<10 MW) (Santos et al. 2012).

Although the downstream effects of large dams on freshwater biota have been well investigated (e.g. Garnier et al. 2000; Gunkel et al. 2003; Xue et al. 2006), little attention has been paid to the effects of small hydropower plants. Although weirs of small hydropower plants are constructed in small and medium-size rivers, their water diversions provoke an alteration of the natural flow regime, with low flows in the reaches below weirs and hydropeaking downstream of the tailrace outlets (Santos et al. 2006; Schmutz et al. 2010; Rolls et al. 2012). The flow regime is viewed as a major factor governing stream ecology (Poff & Ward 1990), so its alteration may provoke significant influences on biota, ecosystem conditions and processes (Lake 2003). The main effects detected on freshwater fishes, in previous studies focusing on low flow due to water diversion of small hydropower plants, were reductions on density and biomass in impacted sites (Kubečka et al. 1997; Almodóvar & Nicola 1999; Mueller et al. 2011). Changes in species composition were also reported in some studies (Mueller et al. 2011), and Anderson et al. (2006) detected an increase in opportunistic species at impacted reaches in Central America. Similarly, Kubečka et al. (1997) showed that the dominance of brown trout (*Salmo trutta*) and other rather large-bodied fish was greatly reduced at impacted reaches and the fish community shifted to a system dominated by small-bodied species in Czech streams. On the other hand, in Mediterranean rivers of Portugal and Spain, significant changes in fish abundance (Anderson et al. 2006; Santos et al. 2006) or species composition were not detected (Almodóvar & Nicola 1999; Santos et al. 2006).

Most of these previous studies have focused on the population and community levels and did not assess effects on individual features. It is widely known that exposure to environmental stressors causes the detrimental effects on important individual fish features such as metabolism, growth, resistance to diseases, reproductive potential and, ultimately, the health, condition and survival of fish (Rice 2001; Barton et al. 2002; Toft et al. 2004). Depending on the intensity and duration of stress exposure and species-specific features, these negative effects may be transferred from the individual to population or community levels (Adams & Greeley 2000). Therefore, studies integrating different fish organisation levels, from individual to community, are needed to understand the impact of water diversion of small hydropower plants.

In this study, we examine the effects of water diversion of small hydropower plants on fish populations

in headwater stream, which has good water quality but a long series of small hydropower plants one after another. We compared control reaches (unimpacted for water diversion) with impacted reaches (downstream of weirs and impacted by low flows but upstream of the tailrace outlets and so not impacted by hydropeaking) in a Pyrenean stream. By comparing close-by control and impacted reaches, we aimed to assess the effects of water diversion of small hydropower plants on habitat features and fishes at three organisation levels (individual, population and community). Pyrenean streams are adequate study systems for this aim because they lack many other perturbations (large dams, significant water pollution or habitat degradation) much more intense in lowland reaches.

Methods

Study area

Sampling was conducted from July to September 2010 at upstream reaches of the Ter River (Catalonia, NE, Spain) (Fig. 1). The study area included the mainstream of Ter River and six tributaries: Rigard, Freser, Segadell, Ritort, Feitús and Riera Major (Fig. 1). The river basin is subject to a Mediterranean climate, although the headwaters are partially subject to a snow-fed regime (Boix et al. 2010). The Ter River originates in the Pyrenees Mountains and has a total drainage area of 2955 km², a mean annual water yield of 845 million m³ and a mean discharge of ca. 10 m³·s⁻¹ (Benejam et al. 2010a). The Ter River and

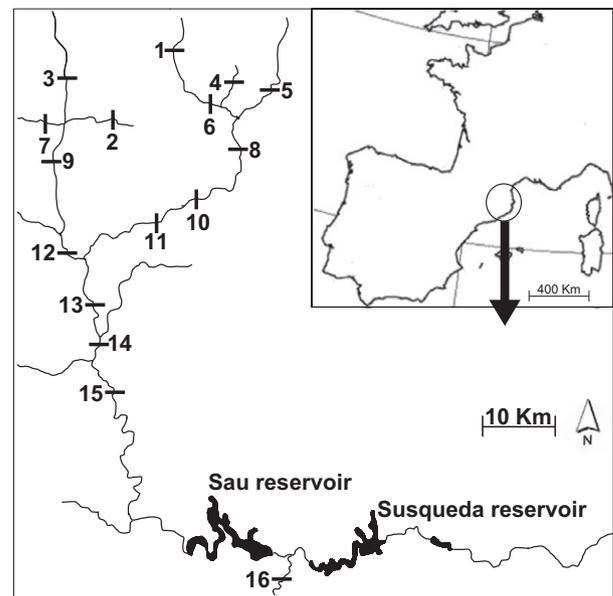


Fig. 1. Study area in the Ter River basin with the 16 hydropower plants studied (see sites codes in Table 1 for further details).

its flow regime are severely affected by two big dams (Sau and Susqueda reservoirs) located in the mid watercourse, which is thus split in the upper Ter basin (with a drainage area of 1799 km²) and the lower basin (with a drainage area of 1156 km²). Our sampling only considered the upper Ter basin, where during the 19th century dozens of small hydropower plants were constructed, as in many rivers of the Pyrenees, to produce energy for textile industries. Nowadays, although factories have closed, there are around 85 small hydropower plants in the upper Ter basin which are still operating to produce energy for the electrical grid (ACA 2010). In this study, 16 of these hydropower plants were studied (Fig. 1, Table 1), only two of them with fish pass facilities. We chose those hydropower plants that had previous data (streamflow of Table 1) and had good access for sampling. The water diversion of these hydropower plants causes an important reduction of streamflow below the weirs (Table 1; ACA 2010). The average features of these plants were 3.4 m of weir height (range of 1.1–7.0 m), 0.789 MW of power (0.02–2.8 MW) and 2 km length of the reach (0.5–5 km) affected by water diversion with reduction of streamflow. The region is rural and sparsely populated (26,393 residents living in an area of 956 km²); the riparian habitat is well preserved and the predominant land cover is forest, with some agricultural and urbanised areas (Benejam et al. 2012). At half of the sampling reaches, brown trout was the only fish species present. In the other sampling reaches, in addition to brown trout, the fish assemblage consisted of Mediterranean barbel (*Barbus meridionalis*) and three non-native species: stone loach (*Barbatula quignardi*), Pyrenean minnow (*Phoxinus phoxinus*) and common carp (*Cyprinus carpio*). Although brown

trout is native to this river basin, many stockings have taken place during the past century and these populations are introgressed with foreign genetic stocks (Sanz et al. 2002). Nowadays, the study area is officially a so-called genetic reserve, and stocking is not allowed.

Experimental design and field methods

In this study, 36 reaches were sampled to study the effect of water diversion of 16 small hydropower plants (Fig. 1, Tables 1 and 2). Each site with one small hydropower plant was sampled upstream (hereafter, control reach) and downstream (hereafter, impacted reach or below impact) of the weir, where water diversion begins. Sampling reaches were at least at a distance of 200 m upstream or downstream to the weir. Impacted reaches were always below the weir but upstream of the tailrace outlet where water is returned after electricity production to estimate the effects of water withdrawal and avoid the effects of hydropeaking. Control reaches were always out of the direct influence of the weir to provide more natural stream features and avoid direct differences in habitat and flow. As weirs of hydropower plants are consecutively situated along the river, we also aimed at analysing the possible cumulative effect along the river, using site as an additional blocking factor to control for longitudinal variation (see details in *Statistical analyses*). In general, two sampling reaches (one control and one impacted reach) were sampled in each site. In some sites (when the length reaches were sufficiently long and the access at sampling site was possible), an additional reach was sampled to increase sample size and statistical power (see cases on Table 2). In these cases, the additional sampling

Table 1. Comparison of the upstream daily flow and flow diversion rates for each hydropower derivation point. Streamflow variables were calculated with 12 years of data series (1997–2009) from gauging stations. NA: in the case of Crous, there are no gauging data available.

| Site code | Name of the hydropower plant | UTM coordinates (31 T) | Upstream maximum daily flow m ³ per s | Upstream average daily flow m ³ per s | Maximum legal flow diversion m ³ per s | Downstream average % daily flow diversion m ³ per s |
|-----------|------------------------------|------------------------|--|--|---|--|
| 1 | Brutau 2 | 440621, 4679727 | 18.5 | 0.7 | 2.0 | 66 |
| 2 | Pardines | 435932, 4685451 | 8.6 | 0.2 | 0.6 | 99 |
| 3 | El Molí Rialp | 431581, 4684282 | 37.8 | 0.7 | 1.8 | 77 |
| 4 | Feitús | 447209, 4688528 | 18.7 | 0.4 | 1.8 | 86 |
| 5 | Cruanyes | 450723, 4687413 | 16.6 | 0.6 | 0.5 | 56 |
| 6 | Brutau 1 | 445846, 4685673 | 46.8 | 1.7 | 1.5 | 76 |
| 7 | Molí de Sart | 430470, 4684704 | 54.4 | 1.0 | 0.8 | 71 |
| 8 | Matabosch | 447302, 4683230 | 81.9 | 3.0 | 2.5 | 76 |
| 9 | Montagut | 430915, 4682486 | 142.8 | 2.7 | 3.0 | 83 |
| 10 | Molí Gran Pont Vell | 442582, 4676858 | 101.5 | 3.6 | 2.5 | 51 |
| 11 | Cal Gat | 440117, 4675454 | 116.2 | 4.3 | 3.0 | 74 |
| 12 | Surribes | 431602, 4673434 | 211.8 | 4.1 | 3.0 | 27 |
| 13 | L'Escala | 434583, 4668204 | 269.7 | 8.7 | 6.0 | 77 |
| 14 | La Cubia | 434583, 4666412 | 269.7 | 8.8 | 6.0 | 77 |
| 15 | Fàbrica Tomàs | 434916, 4662331 | 269.7 | 10.0 | 8.0 | 81 |
| 16 | Crous | 451449, 4643648 | NA | NA | 10.0 | NA |

Table 2. Location, altitude and additional information of sampling reaches, including if they were control reach or impacted reach of water diversion of small hydropower plants and whether three-pass removals with block nets were applied (removal with 'yes').

| Site code | Name of the hydropower plant | Date (dd/mm/yyyy) | UTM coordinates (31 T) | Altitude (m a.s.l.) | Control/impacted reaches | Removal |
|-----------|------------------------------|-------------------|------------------------|---------------------|--------------------------|---------|
| 1 | Brutau 2 | 30/07/2010 | 442652, 4693171 | 1340 | Control | No |
| 1 | Brutau 2 | 25/08/2010 | 442601, 4691697 | 1249 | Control | Yes |
| 1 | Brutau 2 | 23/08/2010 | 442413, 4691523 | 1226 | Impacted | Yes |
| 2 | Pardines | 10/08/2010 | 432702, 4684893 | 951 | Control | Yes |
| 2 | Pardines | 20/08/2010 | 436314, 4684974 | 1173 | Impacted | Yes |
| 2 | Pardines | 10/08/2010 | 434312, 4685083 | 1030 | Impacted | Yes |
| 3 | El Molí Rialp | 11/08/2010 | 431800, 4689717 | 1112 | Control | Yes |
| 3 | El Molí Rialp | 11/08/2010 | 431661, 4689512 | 1100 | Impacted | Yes |
| 3 | El Molí Rialp | 12/08/2010 | 431417, 4688476 | 1050 | Impacted | Yes |
| 4 | Feitús | 09/08/2010 | 447187, 4688551 | 1093 | Control | Yes |
| 4 | Feitús | 29/07/2010 | 447120, 4688377 | 1085 | Impacted | Yes |
| 5 | Cruanyes | 30/07/2010 | 450713, 4687465 | 1010 | Control | Yes |
| 5 | Cruanyes | 29/07/2010 | 448573, 4685254 | 950 | Control | No |
| 5 | Cruanyes | 25/08/2010 | 450791, 4687331 | 1004 | Impacted | Yes |
| 6 | Brutau 1 | 22/09/2010 | 447293, 4685074 | 940 | Control | Yes |
| 6 | Brutau 1 | 24/08/2010 | 446066, 4685636 | 962 | Impacted | Yes |
| 7 | Molí de Sart | 16/08/2010 | 430332, 4684701 | 951 | Control | Yes |
| 7 | Molí de Sart | 16/08/2010 | 430584, 4684641 | 920 | Impacted | Yes |
| 8 | Matabosch | 21/09/2010 | 447582, 4682361 | 900 | Control | No |
| 8 | Matabosch | 21/09/2010 | 447503, 4682556 | 902 | Impacted | No |
| 9 | Montagut | 20/09/2010 | 430973, 4682672 | 860 | Control | No |
| 9 | Montagut | 20/08/2010 | 430836, 4681029 | 840 | Impacted | No |
| 10 | Molí Gran Pont Vell | 22/09/2010 | 442805, 4677203 | 775 | Control | No |
| 10 | Molí Gran Pont Vell | 09/08/2010 | 441626, 4676774 | 758 | Impacted | No |
| 11 | Cal Gat | 23/09/2010 | 440413, 4675459 | 740 | Control | No |
| 11 | Cal Gat | 23/09/2010 | 440017, 4675602 | 730 | Impacted | No |
| 12 | Surribes | 20/09/2010 | 432883, 4672594 | 680 | Control | No |
| 12 | Surribes | 23/08/2010 | 431830, 4673131 | 690 | Impacted | No |
| 13 | L'Escala | 27/09/2010 | 434601, 4668471 | 640 | Control | No |
| 13 | L'Escala | 27/09/2010 | 434494, 4667601 | 630 | Impacted | No |
| 14 | La Cubia | 28/09/2010 | 434567, 4665242 | 610 | Control | No |
| 14 | La Cubia | 28/09/2010 | 434521, 4666201 | 615 | Impacted | No |
| 15 | Fàbrica Tomàs | 27/09/2010 | 434674, 4662704 | 570 | Control | No |
| 15 | Fàbrica Tomàs | 28/09/2010 | 434867, 4662116 | 560 | Impacted | No |
| 16 | Crous | 25/10/2010 | 451197, 4642860 | 445 | Control | No |
| 16 | Crous | 25/10/2010 | 451764, 4643893 | 420 | Impacted | No |

reach was located at least 1.5 km apart from the other reach to insure statistical independence.

Fish were sampled by electrofishing 100-m stretches (200–350 V, 2–3 A, fully rectified triphasic DC). Fishes stunned were collected with nets, identified to species, counted, measured (fork length in mm and total weight in 0.1 g) and then returned to the same reach. Following the CEN standard (CEN 2003), a single electrofishing pass without block nets was applied in general (Table 2). This method has been shown adequate to estimate species richness, species composition and fish abundance in some of the same tributaries of this study (Benejam et al. 2012). At ten sampling reaches, three-pass removals with block nets were applied to estimate population size and capture probability for each species (Table 2).

Several habitat variables were measured at each sampling reach: wetted width and depth (cm), geomorphology (percentage of pools, runs and riffles), substrate composition (percentages of boulder, cobble,

gravel, sand and silt) and percentage of refuges (e.g. large boulders, wood and tree roots). Moreover, a modification (Sostoa et al. 2010) of the Rapid Bioassessment Protocol (RBA) (Barbour et al. 1999) was also applied at all sampling reaches; RBA scores variables such as habitat complexity, mesohabitat diversity, siltation, streamflow, sinuosity, margin stability and aquatic vegetation were calculated. The Riparian Habitat Quality Index (QBR index), which is officially used in the study region, was also estimated at all sampling reaches (Munné et al. 2003). The QBR index is a habitat quality index for riverine areas which considers riparian cover percentage, cover structure and cover quality of the river channel.

Statistical analyses

In the analyses, we aimed to detect changes on numerous fish attributes (from individual condition to species composition) and habitat features due to effects of water diversion of small hydropower

plants, comparing control and impacted reaches (categorical factor). We used site as an additional blocking factor to control for longitudinal variation of the fish assemblage (see Table 1 for site code). We used generalised linear models (GLMs) with Poisson errors and log link functions to test for effects on fish abundance. We considered two measures of fish abundance: population size and catch per unit effort (CPUE). CPUE data were the total fish captured per 100-m stretch in a single electrofishing pass without block nets. We obtained CPUE data (fish/100 m) of all sampling reaches. Population size estimates were only possible at the ten sampling reaches where multiple passes were applied (Table 2) and for these we used program MARK 5.1 (White & Burnham 1999; freely available at <http://warnercnr.colostate.edu/~gwhite/mark/mark.htm>). We estimated population size using the multinomial model ('Huggins Closed Capture' in MARK), with recapture parameter *c* set at zero and constant catchability between different electrofishing passes (Cooch & White 2010). To describe the relationship between trout CPUE with altitude, we used quadratic regression because of clear nonlinear variation (see Carmona-Catot et al. 2010 for a similar pattern and approach).

To test whether control and impacted reaches differed in species composition, we used permutational multivariate analysis of variance using distance matrices (Anderson 2001), as implemented in function 'adonis' of 'vegan' (Oksanen et al. 2012), with Bray-Curtis distances and 999 permutations. 'adonis' seems to be less sensitive to dispersion effects than other more popular alternatives, such as ANOSIM, SIMPER and the multiple response permutation procedure (Oksanen et al. 2012). Because 'adonis' can anyway confound location and dispersion effects, we also tested for the latter using a permutation test of multivariate homogeneity of groups dispersions (Anderson 2006), as available in the function 'permutest.betadisper'. Both tests were performed constraining the permutations with sites, using the 'strata' option in 'vegan'. These analyses were computed with the 'vegan' package (Oksanen et al. 2012) in R (R Development Core Team 2012).

Two-way ANOVAs, with site as a blocking factor to control for longitudinal variation, were used to compare the average of length and total weight of fish between control and impacted reaches. Length and total weight variables were log-transformed because homoscedasticity and linearity were clearly improved. Analysis of covariance (ANCOVA) was used to compare the condition (total weight-length relationship) between control and impacted reaches to impact taking into account fish size (covariate). ANCOVA has several advantages over condition factors (e.g. weight length⁻³) and similar indices (see

reviews in García-Berthou & Moreno-Amich 1993; García-Berthou 2001). The adjusted or predicted means in ANCOVA are the means of values of the response variable adjusted for effects of covariates, typically length (García-Berthou & Moreno-Amich 1993); these adjusted means thus allow comparing groups or treatments for the response variable, after accounting for the effects of fish size.

To compare habitat features between control and impacted reaches, we used two-way ANOVA, with site as an additional factor to control for longitudinal variation. Moreover, multivariate analysis of variance (MANOVA) was also applied to the set of habitat features variables, to control for the overall rate of type I error. To reduce problems with analysing proportional composition data, some categories (e.g. boulder + cobble) were pooled and redundant variables (adding to 100%) were excluded from the analyses. These statistical analyses were performed with SPSS 15 (SPSS Inc., Chicago, IL, USA, 1989–2006).

Results

Effects of small hydropower plants on habitat features

There were significant differences in habitat features between control and impacted reaches for water diversion of small hydropower plants (Tables 3 and 4). In particular, impacted reaches were characterised by shallower water depth, lower presence of total refuges for fish, lower abundance of riffles and higher abundance of pools. (Table 3). The substrate composition was only marginally significant, with a trend of higher presence of boulders and lower of cobbles in the control reaches.

Moreover, the results of Rapid Bioassessment Protocol variables showed that impacted reaches had poorer habitat structure and lower presence of macrophytes (Table 4). The Rapid Bioassessment Protocol variables showed significant differences among sites

Table 3. Two-way ANOVAs of habitat features, with impact and site as factors (d.f. were 1 and 4 for all *F* statistics). Mean values of habitat features for control and impacted reaches are also given. QBR index is a Riparian Habitat Quality Index (Munné et al. 2003).

| Habitat features | <i>F</i> | <i>P</i> value | Control reaches | Impacted reaches |
|--------------------|----------|----------------|-----------------|------------------|
| QBR | 1.80 | 0.25 | 79.69 | 76.88 |
| % riffles | 16.64 | 0.02 | 34.06 | 18.06 |
| % pools | 7.39 | 0.05 | 14.06 | 25.34 |
| % boulder | 4.75 | 0.09 | 32.03 | 24.69 |
| % cobble | 1.01 | 0.37 | 36.09 | 39.84 |
| Total refuges % | 8.07 | 0.05 | 77.50 | 70.62 |
| Maximum depth (cm) | 21.07 | 0.01 | 53.92 | 39.47 |

Table 4. Two-way ANOVAS of 'Rapid Bioassessment Protocol' variables, with impact and site as factors (d.f. were 1 and 4 for all F statistics). R (right side) and L (left side). Mean values of habitat features for control and impacted reaches are also given. All variables potentially range from 1 to 10.

| 'Rapid Bioassessment Protocol' variables | F | P value | Control reaches | Impacted reaches |
|--|------|-----------|-----------------|------------------|
| Habitat structure | 9.80 | 0.04 | 9.25 | 8.59 |
| Habitat diversity | 0.03 | 0.87 | 7.38 | 7.25 |
| Channelling | 0.01 | 0.91 | 8.56 | 8.50 |
| Channel morphology | 0.60 | 0.48 | 8.22 | 7.94 |
| Streamflow | 6.88 | 0.06 | 8.84 | 7.66 |
| Margin erosion R | 4.80 | 0.09 | 7.56 | 7.19 |
| Margin erosion L | 4.80 | 0.09 | 7.56 | 7.19 |
| Macrophytes | 8.07 | 0.05 | 2.41 | 2.06 |
| Riparian vegetation R | 0.24 | 0.65 | 8.06 | 7.88 |
| Riparian vegetation L | 0.24 | 0.65 | 8.06 | 7.88 |
| Width of riparian vegetation R | 0.00 | 0.96 | 7.66 | 7.69 |
| Width of riparian vegetation L | 0.19 | 0.68 | 7.50 | 7.75 |

in the set of variables (MANOVA Wilks' λ : $F_{60, 6.13} = 9.13$; $P = 0.005$) and significant control-impact \times site interaction (MANOVA Wilks' λ : $F_{60, 6.13} = 5.86$; $P = 0.015$), indicating that the magnitude of impacts varied among sites.

Effects of small hydropower plants on fish populations

Fish abundance was higher at control than at impacted reaches for water diversion of small hydropower plants (Fig. 2). All species but common carp had significant site \times impact interactions for catch per unit effort (CPUE) (GLM: Wald chi-square, all $P < 0.01$), suggesting that the impact of water diversion on abundance varies along the stream. Brown trout (GLM: Wald chi-square = 9.86, d.f. = 1, $P = 0.002$) and Pyrenean minnow (GLM: Wald chi-square = 5.72, d.f. = 1, $P = 0.017$) also had significant overall differences in CPUE between control and impacted reaches. The population size of brown trout, estimated using removal data multiple-pass and program MARK, was also higher at control reaches (GLM: Wald chi-square = 71.5, d.f. = 1, $P < 0.0005$) and displayed significant site \times impact interaction (Wald chi-square = 60.3, d.f. = 1, $P < 0.0005$). Brown trout abundance peaked at intermediate altitudes, but was lower at impacted reaches after controlling for this natural variation (Fig. 3). Moreover, the difference in brown trout abundance between control and impacted reaches slightly decreased along the river, from upstream to downstream (Fig. 3).

The average fork length (two-way ANOVA: $F_{1, 34} = 7.25$; $P = 0.011$) and total weight (two-way ANOVA: $F_{1, 34} = 8.43$; $P = 0.006$) of brown trout were significantly higher at control than impacted reaches (Fig. 2). Fish condition (weight after account-

ing for length) of brown trout was also significantly different between control and impacted reaches for hydropower plants (Fig. 4). After accounting for fish size (covariate in ANCOVA), condition of brown trout was lower in impacted reaches (ANCOVA: $F_{1, 1215} = 16.73$; $P < 0.005$) (Fig. 4). This difference in fish condition between control and impacted reaches increased downstream (Fig. 4).

Control and impacted reaches also differed in relative species abundance ('adonis' function, $P = 0.043$). The 'permutest.betadisper' function showed that dispersions were not significantly different ($P = 0.83$), supporting the differences in average species composition. In the impacted reaches, the relative abundance of trout and minnow decreased and that of loach and barbel increased; common carp was only present in two of the sampled impacted reaches. See Appendix S1 for further information.

Discussion

Our results indicate that water diversion of small hydropower plants is affecting the habitat features and fish assemblages in Pyrenean streams. In the impacted reaches, we have shown a significant lower presence of total refuges for fish, poorer structure of habitat, lower frequency of riffles but higher of pools, shallower water levels and lower presence of macrophytes. We also detected lower values of fish abundance, average fork length, total weight and fish condition at impacted reaches. Moreover, species composition was also affected with lower relative abundance of trout and minnow at impacted reaches and higher presence of loach and barbel.

The detected impacts of low flow due to water diversion of small hydropower plants on habitat features agree with previous studies in contrasting regions (Kubečka et al. 1997; Anderson et al. 2006; Wu et al. 2009; Mueller et al. 2011). River discharge controls the diversity and availability of habitats, such as riffles, runs, pools, backwaters and floodplains. Low flows reduce volume, area and depth of aquatic habitat and change the instantaneous velocity of rivers (Rolls et al. 2012). For example, in headwater streams of West Virginia, USA, a 96% reduction in discharge resulted in a 52% reduction of riffle habitat area (Hakala & Hartman 2004). Anderson et al. (2006) observed similar results in the Puerto Viejo River, Costa Rica, where a larger percentage of the channel consisted of rapids and riffle habitats upstream from weirs of small hydropower plants, whereas pool or low water velocity habitats dominated at downstream, dewatered reaches. Comparable results were detected in different rivers of the Czech Republic, with higher abundance of pools and lower depth, water velocity and wetted surface area at sites

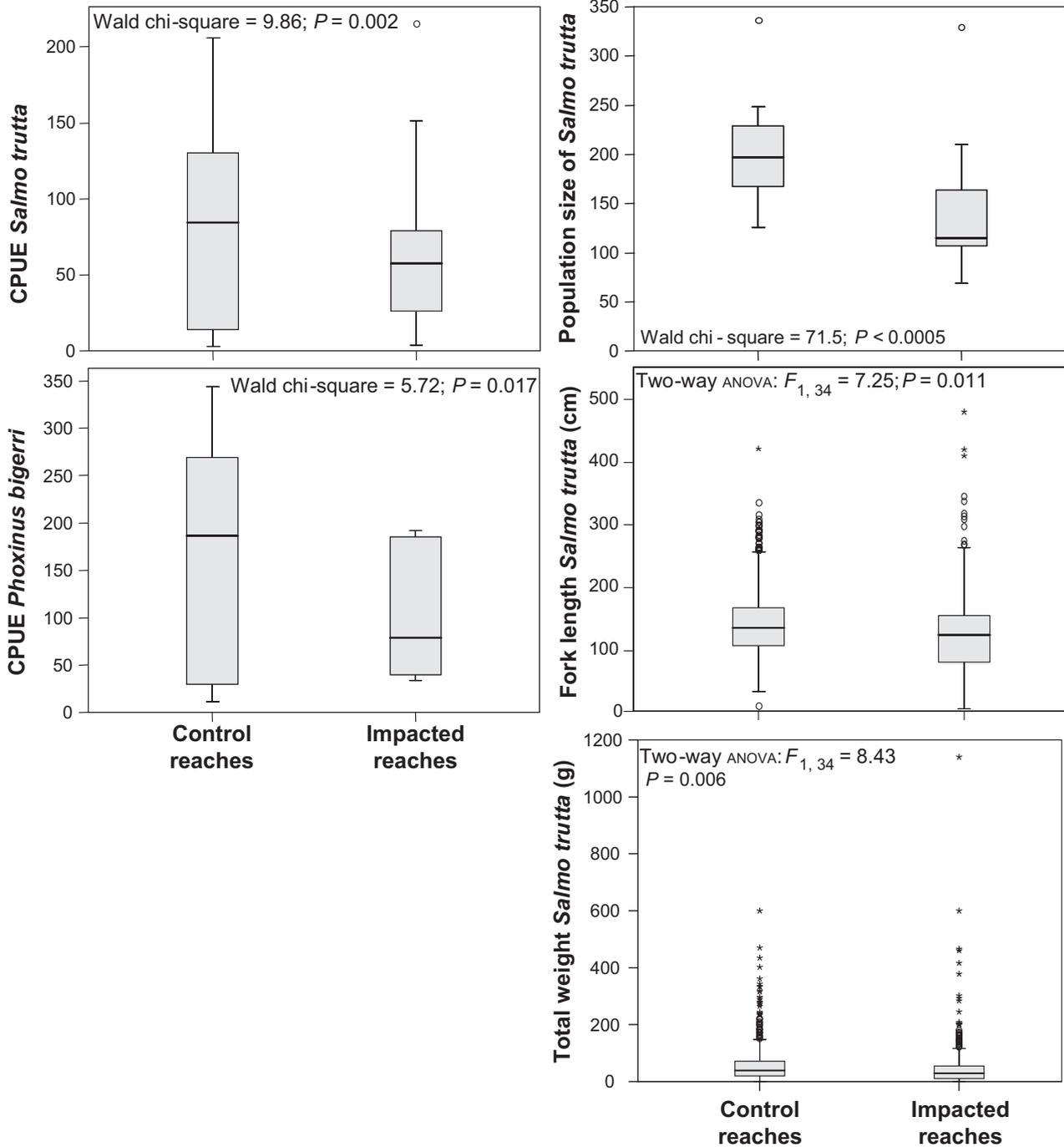


Fig. 2. Box-plots of fish abundances and fork lengths at control and impacted reaches. Only species with significant results are shown. Box corresponds to 25th and 75th percentiles; dark line inside the box represents the median; error bars show the minimum and maximum except for outliers (open circles and asterisks, corresponding to values beyond 1.5 and 3 box lengths, respectively, from the box).

impacted by water diversion of small hydropower plants (Kubečka et al. 1997). The overall differences detected in our study on habitat features between control and impacted reaches may be linked to changes on fish assemblages.

Brown trout was affected by the small hydropower plants analysed, with smaller mean size and lower abundance and condition at impacted reaches. Moreover, the cumulative impact of hydropower plants

along the river was detected because the difference in condition between control and impacted reaches increased downstream. The condition or health of fish individuals is a measure of the physical and biological circumstances during recent life and is affected by interactions among food availability, physical factors and environmental conditions (Lloret & Rätz 2000; Vila-Gispert et al. 2000). Fish condition is particularly important because it has strong influences

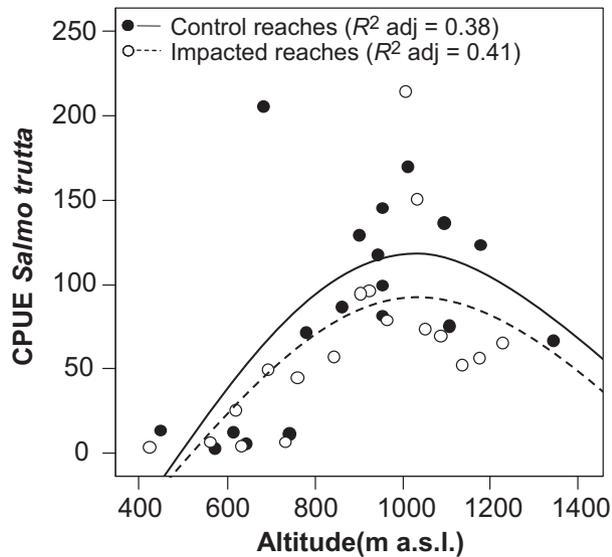


Fig. 3. Distribution of brown trout abundance (CPUE) by sampling sites ordered by altitude. Quadratic regressions per reach type (control and impacted) are also shown.

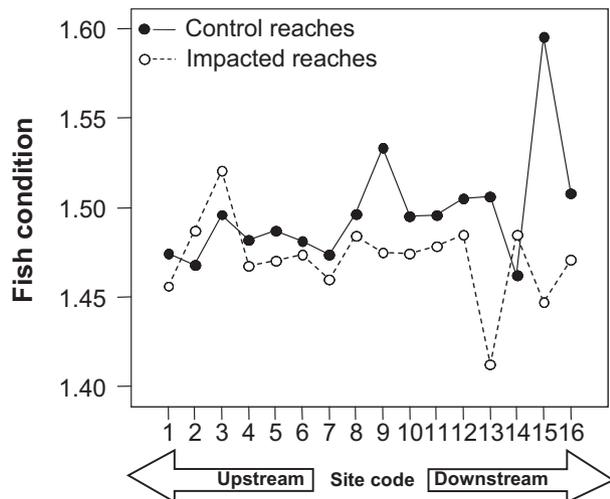


Fig. 4. Comparison of the condition of brown trout along sites between control and impacted reaches. Size-adjusted means (ANCOVA with fish length as covariate) of total weight are shown. Increasing site code indicates downstream direction (see Table 1).

on growth, reproduction and survival of individuals and thus affects other ecological levels, such as populations and communities (Lambert & Dutil 1997; Adams 1999; Marshall & Frank 1999). Although many studies have shown results of low fish condition in ecosystems with poor water quality in a range of species (Laflamme et al. 2000; Oliva-Paterna et al. 2003; Benejam et al. 2010b), as far as we know only Torralva et al. (1997) reported changes in fish condition almost entirely due to flow regime alteration. However, Torralva et al. (1997) studied the impact of two big reservoirs, not small hydropower plants, and they observed differences in condition but were not able to statistically test them.

The lower abundance of fish at impacted reaches detected in our study agrees with the results of other authors, who in some cases found four times higher biomass at control reaches (Kubečka et al. 1997; Almodóvar & Nicola 1999). Limited food resources and loss of preferred habitat have been attributed as causes of reduced densities of fish in sites affected by low flows (Hakala & Hartman 2004; Riley et al. 2009). Concretely, Lobón-Cerviá (2009, 2013) detected that low discharge and shallow water levels in March induced lower recruitment and survival rates of brown trout because it affected the emergence period and the earliest search stages for food. Nicola et al. (2009) also showed that the magnitude and duration of low flows during summer drought appeared to be a critical factor for survival of young brown trout.

Population size structure is considered a good health indicator in freshwater bioassessment, because it has the potential to inform us on whether disturbance is affecting populations (Karr et al. 1986; Murphy et al. 2013). Body size is a fundamental characteristic of organisms and arguably the most important trait affecting the ecological performance of individuals (Persson & de Roos 2007; Murphy et al. 2013). The implications of body size on growth, mortality and trophic interactions highlight the importance of size structure for populations (de Roos et al. 2003; Savage et al. 2004; Brown et al. 2007). In our study, we detected lower values of average fork length and total weight for brown trout at impacted reaches. Decreased average sizes have also been reported in various taxa as a response to anthropogenic perturbations (Dodson & Hanazato 1995; Jung & Jagoe 1995; Walters & Post 2008). In the same region of our study, Murphy et al. (2013) found opposite response of size structure for chub (*Squalius laietanus*), with increasing average size of populations under increasingly disturbed conditions. The authors suggest that this unusual pattern may reflect failure to recruit in disturbed conditions or growth at reduced densities, as increases in length appear to be related to a trend of decreasing abundance. However, in our study, fish abundance was higher in control sites where average fork length and total weight also increased, which seems a more expected response.

Overall, our results show that the response to environmental perturbation due to water diversion of small hydropower plants was species specific and brown trout was the species with the clearest effects on fitness-related traits at impacted reaches. It is widely known that brown trout is an intolerant species of poor water quality and habitat structure (Blanco & González 1992; Maceda-Veiga & Sostoa 2011). Consequently, brown trout seems more

affected by small hydropower plants than Mediterranean barbel, stone loach or Pyrenean minnow. As a result, species composition was affected by water diversion of small hydropower plants with lower relative abundance of trout and minnow at impacted reaches and higher presence of loach and barbel. The alteration of natural flow regime provides a habitat more suitable to generalist and opportunistic faunas, giving these species a competitive advantage (Poff & Ward 1989).

Although the effects of water diversion of small hydropower plants on fish populations and habitat features that we detected seem logical, the impact may be underestimated because control reaches were not pristine sites and the impacts accumulate along streams. The weirs of hydropower plants are consecutively situated along the river, compromising connectivity; therefore, their impact accumulates downstream and many control sites have previous impacts upstream, as seen in our condition fish data and CPUE for trout. It is likely that other cumulative effects not detected in our study are occurring.

Adams & Greeley (2000) showed that the transfer of effects from individual to community levels depends on the intensity and duration of stress exposure. In our study, we detected effects of water diversion of small hydropower plants on fish assemblages at the three main organisation levels in ecology: individual (fish condition), population (abundance, average length and weight) and community (relative abundance of fish species). Therefore, despite the small size of these hydropower plants in headwater streams, their impact seems quite important, and these fish metrics could be used to evaluate ecosystem health during mitigation or restoration activities.

Some management tools might mitigate the negative effects of small hydropower plants. It is widely known that dams and weirs interrupt the longitudinal river connectivity, isolating fish communities (Santos et al. 2006). Longitudinal river connectivity is a basic requirement for fish community persistence, as it allows seasonal movements, enhances lifetime reproductive success and allows recolonisation of areas affected by disturbance (Ordeix et al. 2011). Therefore, the presence of effective fish passes in all weirs would allow connectivity along the river and would decrease their impact. However, recreating environmental flows as similar as possible to the natural flow regime is the main tool to reduce the impact of small hydropower plants, because flow regime is the major factor governing stream ecology (Poff & Ward 1990; Lake 2003). As a result of the requirements of the Water Framework Directive (WFD) (2000/60/CE), many European countries implement new regulations to restore the natural flow regime and improve ecological status of freshwater ecosystems. Many of the

hydropower plants have old water rights, most of which expire in 2061, with few environmental obligations. Therefore, it is essential that hydropower plants apply environmental flows and other measures such as building effective fish passes to mitigate their impact on the freshwater biota and ecosystem services to achieve good ecological status according to the WFD requirements. Additionally, water authorities need to improve biological indices and suitable monitoring programs to properly detect flow regime alteration produced by hydropower plants, especially in clean waters and near natural headwaters where chemical quality is mainly good.

Acknowledgements

We thank A. Saperas, J. Guillaumet, E. Navarro and A. Vilajeliu for assistance in field work and anonymous reviewers for helpful comments. This study was financed by the Catalan Water Agency (ACA), Government of Catalonia (CTN1001569). Additional financial support was provided by the Spanish Ministry of Economy and Competitiveness (project CGL2013-43822-R) and the Government of Catalonia (ref. 2014 SGR 484).

References

- ACA (Agència Catalana de l'Aigua). 2010. Report of environmental sustainability the zonal plan for implementing environmental flows in the upper basin of Ter River. Barcelona: Generalitat de Catalunya. Available at: http://aca-web.gencat.cat/aca/documents/ca/legislacio/projectes/12_IS_Pla_zonal_manteniment_Ter_superior.pdf. Last accessed 16 July 2014.
- Adams, S.M. 1999. Ecological role of lipids in the health and success of fish populations. In: Arts, M.T. & Wainman, B.C., eds. The ecological role of lipids in freshwater ecosystems. New York: Springer-Verlag, pp. 132–160.
- Adams, S.M. & Greeley, M.S. 2000. Ecotoxicological indicators of water quality: using multi-response indicators to assess the health of aquatic ecosystems. *Water Air and Soil Pollution* 123: 103–115.
- Almodóvar, A. & Nicola, G.G. 1999. Effects of a small hydropower station upon brown trout *Salmo trutta* L. in the River Hoz Seca (Tagus basin, Spain) one year after regulation. *Regulated Rivers: Research & Management*. 15: 477–484.
- Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. *Austral Ecology* 26: 32–46.
- Anderson, M.J. 2006. Distance-based tests for homogeneity of multivariate dispersions. *Biometrics* 62: 245–253.
- Anderson, E.P., Freeman, M.C. & Pringle, C.M. 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.
- Ayllón, D., Almodóvar, A., Nicola, G.G., Parra, I. & Elvira, B. 2012. A new biological indicator to assess the ecological status of Mediterranean trout type streams. *Ecological Indicators* 20: 295–303.

- Barbour, M.T., Gerritsen, J., Snyder, B.D. & Stribling, J.B. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd edn. Washington, D.C.: U.S. Environmental Protection Agency; Office of Water.
- Barton, B.A., Morgan, J.D. & Vijayan, M.M. 2002. Physiological condition-related indicators of environmental stress in fish. In: Adams, S.M., ed. *Biological Indicators of Aquatic Ecosystem Stress*. Bethesda, MD: American Fisheries Society, pp. 111–148.
- Benejam, L., Angermeier, P.L., Munné, A. & García-Berthou, E. 2010a. Assessing effects of water abstraction on fish assemblages in Mediterranean streams. *Freshwater Biology* 55: 628–642.
- Benejam, L., Benito, J. & García-Berthou, E. 2010b. Decrease of condition and fecundity of freshwater fish in a highly polluted reservoir. *Water, Air, & Soil Pollution* 210: 231–242.
- Benejam, L., Alcaraz, C., Benito, J., Caiola, N., Casals, F., Maceda-Veiga, A., Sostoa, A. & García-Berthou, E. 2012. Fish catchability and comparison of four electrofishing crews in Mediterranean streams. *Fisheries Research* 123: 9–15.
- Blanco, J.C. & González, J.L. 1992. *Libro rojo de los vertebrados de España*. Madrid: ICONA.
- Boix, D., García-Berthou, E., Gascón, S., Benejam, L., Tornés, E., Sala, J., Benito, J., Munné, A., Solà, C. & Sabater, S. 2010. Response of community structure to sustained drought in Mediterranean rivers. *Journal of Hydrology* 383: 135–146.
- Brown, J., Allen, A. & Gillooly, J. 2007. The metabolic theory of ecology and the role of body size in marine and freshwater ecosystems. In: Hildrew, A., Raffaelli, D. & Edmonds-Brown, R., eds. *Body size: the structure and function of aquatic ecosystems*. Cambridge: Cambridge University Press, pp. 1–15.
- Carmona-Catot, G., Moyle, P.B., Aparicio, E., Crain, P.K., Thompson, L.C. & García-Berthou, E. 2010. Brook trout removal as a conservation tool to restore Eagle Lake rainbow trout. *North American Journal of Fisheries Management* 30: 1315–1323.
- CEN. 2003. Water quality - sampling of fish with electricity. European standard - EN 14011:2003, Brussels.
- Cooch, E. & White, G. 2010. Program MARK: a gentle introduction. Colorado State University, Fort Collins, Available at: <http://www.phidot.org/software/mark/docs/book/>. Last accessed 16 July 2014.
- de Roos, A., Persson, L. & McCauley, E. 2003. The influence of size-dependent life history traits on the structure and dynamics of populations and communities. *Ecology Letters* 6: 473–487.
- Dodson, S.I. & Hanazato, T. 1995. Commentary on effects of anthropogenic and natural organic chemicals on development, swimming behavior, and reproduction of *Daphnia*, a key member of aquatic ecosystems. *Environmental Health Perspectives* 103: 6917–11.
- Energy Information Administration (EIA). 2011. Annual energy outlook. Washington, DC: U.S. Energy Information Administration. Available at: [http://205.254.135.7/forecasts/aeo/pdf/0383\(2011\).pdf](http://205.254.135.7/forecasts/aeo/pdf/0383(2011).pdf). Last accessed 16 July 2014.
- Espejo, C. & García, R. 2010. Agua y energía: producción hidroeléctrica en España. *Investigaciones Geográficas* 51: 107–129.
- García-Berthou, E. 2001. Size- and depth-dependent variation in habitat and diet of the common carp (*Cyprinus carpio*). *Aquatic Sciences* 63: 466–476.
- García-Berthou, E. & Moreno-Amich, R. 1993. Multivariate analysis of covariance in morphometric studies of the reproductive cycle. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 1394–1399.
- Garnier, J., Billen, G., Sanchez, N. & Leporeq, B. 2000. Ecological functioning of the Marne reservoir (Upper Seine Basin, France). *Regulated Rivers: Research and Management* 16: 51–71.
- Gunkel, G., Lange, U., Walde, D. & Rosa, J.W.C. 2003. The environmental and operational impacts of Curuá-Una, a reservoir in the Amazon region of Pará, Brazil. *Lakes & Reservoirs: Research & Management* 8: 201–216.
- Habit, E., Belk, M.C. & Parra, O. 2007. Response of the riverine fish community to the construction and operation of a diversion hydropower plant in central Chile. *Aquatic Conservation: Marine and Freshwater Ecosystems* 17: 37–49.
- Hakala, J.P. & Hartman, K.J. 2004. Drought effect on stream morphology and brook trout (*Salvelinus fontinalis*) populations in forested headwater streams. *Hydrobiologia* 515: 203–213.
- Jung, R.E. & Jagoe, C.H. 1995. Effects of low pH and aluminum on body size, swimming performance, and susceptibility to predation of green tree frog (*Hyla cinerea*) tadpoles. *Canadian Journal of Zoology* 73: 2171–2183.
- Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R. & Schlosser, I.J. 1986. Assessing biological integrity in running waters: a method and its rationale. Champaign, IL: Illinois Natural History Survey.
- Koç, C. 2012. Problems and solutions related to hydroelectric power plants constructed on the Buyuk Menderes and the West Mediterranean Basin. *Energy Sources, Part A: Recovery, Utilization, and Environmental Effects* 34: 1416–1425.
- Kubečka, J., Matena, J. & Hartvich, P. 1997. Adverse ecological effects of small hydropower stations in the Czech Republic: 1. Bypass plants. *Regulated Rivers: Research & Management* 13: 101–113.
- Laflamme, J.S., Couillard, Y., Campbell, P.G.C. & Hontela, A. 2000. Interrenal metallothionein and cortisol secretion in relation to Cd, Cu, and Zn exposure in yellow perch, *Perca flavescens*, from Abitibi lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 57: 1692–1700.
- Lake, P.S. 2003. Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology* 48: 1161–1172.
- Lambert, Y. & Dutil, J.D. 1997. Can simple condition indices be used to monitor and quantify seasonal changes in the energy reserves of Atlantic cod (*Gadus morhua*). *Canadian Journal of Fisheries and Aquatic Sciences* 54: 104–112.
- Loret, J. & Rätz, H.J. 2000. Condition of cod (*Gadus morhua*) off Greenland during 1982–1998. *Fisheries Research* 48: 79–86.
- Lobón-Cerviá, J. 2009. Why, when and how do fish populations decline, collapse and recover? The example of brown trout (*Salmo trutta*) in Rio Chaballos (northwestern Spain). *Freshwater Biology* 54: 1149–1162.
- Lobón-Cerviá, J. 2013. Recruitment and survival rate variability in fish populations: density-dependent regulation or fur-

- ther evidence of environmental determinants? Canadian Journal of Fisheries and Aquatic Sciences 71: 1–11.
- Maceda-Veiga, A. & Sostoa, A. 2011. Observational evidence of the sensitivity of some fish species to environmental stressors in Mediterranean rivers. Ecological Indicators 11: 311–317.
- Marshall, C.T. & Frank, K.T. 1999. The effect of interannual variation in growth and condition on haddock recruitment. Canadian Journal of Fisheries and Aquatic Sciences 56: 347–355.
- Mueller, M., Pander, J. & Geist, J. 2011. The effects of weirs on structural stream habitat and biological communities. Journal of Applied Ecology 48: 1450–1461.
- Munné, A., Prat, N., Sola, C., Bonada, N. & Rieradevall, M. 2003. A simple field method for assessing the ecological quality of riparian habitat in rivers and streams: QBR index. Aquatic Conservation-Marine and Freshwater Ecosystems 13: 147–163.
- Murphy, C.A., Casals, F., Solà, C., Caiola, N., Sostoa, A. & García-Berthou, E. 2013. Efficacy of population size structure as a bioassessment tool in freshwaters. Ecological Indicators 34: 571–579.
- Nicola, G.G., Almodóvar, A. & Elvira, B. 2009. Influence of hydrologic attributes on brown trout recruitment in low-latitude range margins. Oecologia 160: 515–524.
- Nilsson, C., Reidy, C.A., Dynesius, M. & Revenga, C. 2005. Fragmentation and flow regulation of the world's large river systems. Science 308: 405–408.
- Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., O'Hara, R.G., Simpson, G.L., Solymos, P., Stevens, M.H.H. & Wagner, H. 2012. Vegan: Community Ecology Package. R package version 2.0-4. Available at: <http://CRAN.R-project.org/package=vegan>. Last accessed 16 July 2014.
- Oliva-Paterna, F.J., Vila-Gispert, A. & Torralva, M. 2003. Condition of *Barbus sclateri* from semi-arid aquatic systems: effects of habitat quality disturbances. Journal of Fish Biology 63: 699–709.
- Ordeix, M., Pou-Rovira, Q., Sellares, N., Bardina, M., Casamitjana, A., Solà, C. & Munné, A. 2011. Fish pass assessment in the rivers of Catalonia (NE Iberian Peninsula). A case study of weirs associated with hydropower plants and gauging stations. Limnetica 30: 405–426.
- Persson, L. & de Roos, A. 2007. Interplay between individual growth and population feedbacks shapes body-size distributions. In: Hildrew, A., Raffaelli, D. & Edmonds-Brown, R., eds. Body size: the structure and function of aquatic ecosystems. Cambridge: Cambridge University Press, pp. 225–244.
- Pires, A.M., Cowx, I.G. & Coelho, M.M. 1999. Seasonal changes in fish community structure of intermittent streams in the middle reaches of the Guadiana basin, Portugal. Journal of Fish Biology 54: 235–249.
- Poff, N.L. & Ward, J.V. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46: 1805–1818.
- Poff, N.L. & Ward, J.V. 1990. The physical habitat template of lotic systems: recovery in the context of historical pattern of spatio-temporal heterogeneity. Environmental Management 14: 629–646.
- R Development Core Team. 2012. R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.
- Rice, C.D. 2001. Fish immunotoxicology. In: Schlenk, D. & Benson, W., eds. Target organ toxicity in marine and freshwater teleosts. New York: CRC Press. 382 pp.
- Riley, W.D., Maxwell, D.L., Pawson, M.G. & Ives, M.J. 2009. The effects of low summer flow on wild salmon (*Salmo salar*), trout (*Salmo trutta*) and grayling (*Thymallus thymallus*) in a small stream. Freshwater Biology 54: 2581–2599.
- Rolls, R.J., Leigh, C. & Sheldon, F. 2012. Mechanistic effects of low-flow hydrology on riverine ecosystems: ecological principles and consequences of alteration. Freshwater Science 31: 1163–1186.
- Santos, J.M., Ferreira, M.T., Pinheiro, A.N. & Bochechas, J.H. 2006. Effects of small hydropower plants on fish assemblages in medium-sized streams in central and northern Portugal. Aquatic Conservation: Marine and Freshwater Ecosystems 16: 373–388.
- Santos, J.M., Silva, A.T., Katopodis, C., Pinheiro, P.J., Pinheiro, A.N., Bochechas, J. & Ferreira, M.T. 2012. Ecohydraulics of pool-type fishways: getting past the barriers. Ecological Engineering 48: 38–50.
- Sanz, N., García-Marín, J.L. & Pla, C. 2002. Managing fish populations under mosaic relationships. The case of brown trout (*Salmo trutta*) in peripheral Mediterranean populations. Conservation Genetics 3: 385–400.
- Savage, V.M., Gillooly, J.F., Brown, J.H., West, G.B. & Charnov, E.L. 2004. Effects of body size and temperature on population growth. American Naturalist 177: 429–441.
- Schmutz, S., Melcher, A., Jungwirth, M., Schinegger, R., Unfer, G., Wiesner, C. & Zeiringer, B. 2010. Hydro peaking in Austria and its effects on aquatic organisms. In Abstract Book - 8th International Symposium on Ecohydraulics 2010. Bridging between Ecology and Hydraulics and Leading the Society's New Need - Living with Nature.
- Sostoa, A., Maceda, A., García-Berthou, E., Benejam, L., Casals, F., Caiola, N. & Alcaraz, C. 2010. Adjustment of IBICAT (biotic integrity index of Catalonia) based on the use of fish as indicators of the environmental quality of the rivers of Catalonia. Barcelona: Agència Catalana de l'Aigua (ACA), Generalitat de Catalunya. Available at: http://aca-web.gencat.cat/aca/documents/ca/directiva_marc/IBICAT2_Informe2010.pdf. Last accessed 16 July 2014.
- Toft, G., Baatrup, E. & Guillelte, L.J. 2004. Altered social behavior and sexual characteristics in mosquitofish (*Gambusia holbrooki*) living downstream of a paper mill. Aquatic Toxicology 70: 213–222.
- Torralva, M.M., Puig, M.A. & Fernández-Delgado, C. 1997. Effect of river regulation on the life-history patterns of *Barbus sclateri* in the Segura river basin (southeast Spain). Journal of Fish Biology 51: 300–311.
- Vila-Gispert, A., Zamora, L. & Moreno-Amich, R. 2000. Use of the condition of Mediterranean barbel (*Barbus meridionalis*) to assess habitat quality in stream ecosystems. Archiv für Hydrobiologie 148: 135–145.
- Walters, A.W. & Post, D.M. 2008. An experimental disturbance alters fish size structure but not food chain length in streams. Ecology 89: 3261–3267.

Benejam et al.

- White, G.C. & Burnham, K.P. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study* 46: 120–138.
- Wu, N., Tang, T., Zhou, S.C., Jia, X.H., Li, D.F., Liu, R.Q. & Cai, Q.H. 2009. Changes in benthic algal communities following construction of a run-of-river dam. *Journal of the North American Benthological Society* 28: 69–79.
- Xenopoulos, M.A. & Lodge, D.M. 2006. Going with the flow: using species-discharge relationships to forecast losses in fish biodiversity. *Ecology* 87: 1907–1914.
- Xue, J.Z., Ye, L., Cai, Q.H. & Liu, J.K. 2006. Variation of cladocerans from Maoping to Guizhou in the Three Gorges Reservoir before and after impoundment. *Acta Hydrobiologica Sinica* 30: 58–63.
- Zhang, Q., Li, L., Wang, Y.G., Werner, A.D., Xin, P., Jiang, T. & Barry, D.A. 2012. Has the Three-Gorges Dam made the Poyang Lake wetlands wetter and drier? *Geophysical Research Letters* 39: 1–7.

Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Fish individuals captured at each site.