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The impacts of 'run-of-river' hydropower on the physical and ecological condition of rivers

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Abstract

This paper synthesises published literature on run-of-river hydropower, highlighting its potential to affect both the physical and ecological conditions of river systems. The paper considers the limited number of direct studies and reviews a wider literature on the two principal impacts of such schemes on river systems: the introduction or maintenance of in-channel barriers and water abstraction/flow regime alteration. We outline how river systems are likely to be impacted by such schemes and identify the key issues arising from their continued development. Potential mitigation approaches are highlighted and the areas of future research required to adequately address current knowledge gaps are identified.

Introduction

Globally, hydropower accounts for 16% of electricity production (WEC 2010), more than any other renewable source, and recent drivers to expand renewable energy production, such as European Union (EU) legislation requirements for 20% of energy production from renewable sources by 2020 (EPCEU 2009), have increased interest in hydropower. Extraction of this resource has typically involved damming rivers and creating large reservoirs, with well-documented environmental consequences (Baxter 1977; Pringle *et al*. 2000; Poff & Hart 2002). However, as most opportunities for economically profitable medium- to large-scale schemes in Europe have already been developed (Paish 2002), with the remainder considered either environmentally unacceptable or politically unfavourable (Abbasi & Abbasi 2011), attention has turned to smaller-scale hydropower, particularly run-of-river (ROR) schemes, which are widely regarded as less environmentally damaging (Paish 2002; BHA 2005). However, evidence to support this assumption is scarce (Abbasi & Abbasi 2011), although a recent report by Robson *et al*. (2011) applied research from water abstraction and large (dam) hydropower to ROR schemes and suggested potentially significant impacts on fish communities. There is, therefore, an urgent need to review current understanding of the impacts of such schemes. This is particularly pertinent in the UK and Europe, where there has been a surge in hydropower development (SPLASH 2005; Kucukali & Baris 2009), prompted by financial subsidies from EU and national renewable energy legislation, but also a legislative requirement for all waterbodies to reach 'good ecological status' under the EU Water Framework Directive (WFD) (2000/60/EC). However, hydropower expansion is also occurring in parts of Oceania and Asia, and many of the same considerations apply there.

This paper reviews the current science surrounding ROR hydropower schemes, considering their physical (i.e. hydromorphological) and ecological impacts on rivers, and potential mitigation techniques. The pertinent questions for further research are identified, with a focus on the UK and Europe.

ROR hydropower

ROR hydropower schemes operate without water storage, using the flow within a river channel. Channel obstructions (typically weirs) regulate water levels, allowing a proportion of flow to be diverted down a secondary channel to a turbine before it is returned to the main channel further downstream (Fig. 1). Such schemes vary in size; some larger installations located on major rivers can have peak capacities of >1 MW, whereas the smallest ROR schemes on streams have peak capacities of <10 kW. In Europe, the majority of ROR schemes are mini (<1 MW) and micro (<100 kW) schemes installed on smaller river systems, with relatively few multiple megawatt schemes.

ROR schemes vary considerably in design, as they are tailored to the site geography and, importantly within Europe, historical use and modification. However, three basic classifications can be identified: high-head schemes, low-head diversion schemes and low-head in-weir schemes. High-head schemes use relatively small volumes of water, diverted over long distances (typically >1 km) and are confined to

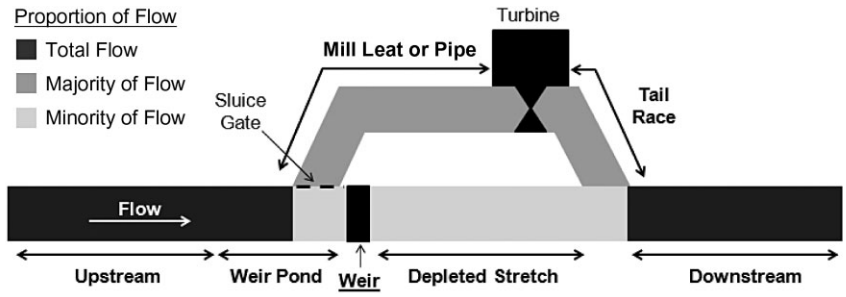

Fig. 1. A schematic representation of a run-of-river hydropower installation, highlighting the key components present for both high- and low-head diversion schemes.

Fig. 2. An example of a low-head in-weir scheme (Torrs Hydro Scheme, New Mills, Derbyshire, UK).

high-gradient, upland rivers. The head (vertical fall) is usually provided by natural waterfalls or cascades, but small (<0.5 m) weirs are still required to divert water. Low-head schemes occur in lower gradient river reaches and are often retrofitted to existing structures. These may be old mill systems, which utilise existing leats (Fig. 1) or schemes installed adjacent to weirs (Fig. 2). In both cases, relatively large volumes of water are diverted, but the distance between diversion and return of the water is typically much greater in the former (0.1–1 km) than the latter (<0.05 km). Additional to these, ROR hydrokinetic schemes use the kinetic energy of a river to drive a turbine or propeller in the channel, without barriers or flow diversions. These occur primarily in the US and are still relatively rare. As such, they are excluded from this review, but further information on such schemes can be found in Khan *et al*. (2009) and NRC (2013).

Power generation for ROR schemes is usually by either 'fast' rotation impulse (high head) or reaction (low head) turbines (BHA 2005). The UK and Europe has seen a surge in slower rotation devices, principally reverse screws, at low-head sites (Bracken & Lucas 2013; Lyons & Lubitz 2013).

Potential physical and ecological impacts

Hydrological and geomorphological interactions within rivers determine the physical habitat, which, in turn, influences biological communities (Poole 2010; Newson *et al*. 2012). However, very few studies have researched this is in the context of ROR hydropower (see Table 1 for a synopsis). Therefore, to gain insight into the potential effects of ROR hydropower, wider research on the ecological impacts of hydrological and geomorphological change in rivers needs to be considered. This is focussed on two key modifications resulting from ROR hydropower (hereafter referred to as hydropower) schemes: in-channel barriers and flow regime change.

Impacts of in-channel barriers

Most high-head schemes require the construction of a new in-channel barrier. Low-head schemes typically utilise existing weirs or other structures, particularly in Europe. While retrofitting does not create a new barrier, it may drive the repair or enhancement of existing structures and remove the feasibility of weir notching or removal (although this practice is not widespread within Europe). The presence of a barrier has two major impacts on a river ecosystem: (1) it disrupts longitudinal connectivity (Vannote *et al*. 1980; Stanford & Ward 2001), fragmenting the river; (2) it alters the in-channel environment and thus physical habitat (Csiki & Rhoads 2010).

The disruption of the longitudinal continuum caused by weirs hinders the natural downstream movement of

Table 1 A summary of the published studies investigating the physical, hydrological and ecological impacts of operational hydropower schemes from around the world

Study site/area	Installed capacity	Physical habitat and hydrological impacts	Ecological impacts	Author
High-head diversion				
10 schemes on rivers around the River Tay Catchment, Scotland	0.68–3.0 MW	• Depleted flows range 5–36% of total flow.	• 20% of schemes exhibited reductions in salmonids in depleted stretches.	Robson (2013)
Afon Ty'n-y-Ddol, Wales	600 kW	• Hydrological modelling suggests almost total removal of midflows in depleted stretch.	• Slight reduction in mayfly abundance in depleted stretch.	Copeman (1997)
Low-head diversion				
Puerto Viejo River (Q_{mean} 8.5 m^3/s) and Quebradon Stream (Q_{mean} 1 m^3/s), Costa Rica	18 MW	• Flow reduction in depleted stretch up to 90% of Q_{mean}. • Water velocities in pools and riffles 4–10 times lower in depleted stretches.	• Fish with involved reproductive requirements, e.g. cichlids, that require parental care, absent in depleted stretches.	Anderson *et al.* (2006)
Laja (low flow Q_{mean}: 47.2 m^3/s) and Rucue Rivers (4.4 m^3/s), Chile	Unknown. Turbine capacity 130 m^3/s	• Flow reduction in depleted stretch of up to 96% (Laja River) and 95% (Rucue River) of average flow.	• Laja River: Notable reduction in non-native salmonid population in depleted stretch following scheme installation. • Rucue River: Total fish abundance reduced in depleted stretch, but fish community structure less impacted because of the presence of additional native species.	Habit *et al.* (2007)
River Lhomme, Belgium (Q_{mean}: 1.78 m^3/s)	Unknown	• Near total loss of deep run habitats in depleted stretch.	• Total reduction in fish biomass of 50–59% in depleted stretch, 4 years after scheme introduction. • Fish population shift from adults to juveniles.	Ovidio *et al.* (2008)
23 diversion schemes in Czech Republic. Q_{mean} 0.08–4.06 m^3/s	Unknown. Turbine capacity 0.011–3 m^3/s	• Flow in depleted stretch not maintained in many cases, causing a reduction in wetted width by 0–50%.	• 36 and 71% of weirs form a migratory barrier for brown trout and grayling, respectively. • General absence of large-bodied fish species, e.g. brown trout, in depleted stretches. • No pronounced macroinvertebrate community differences between depleted and nondepleted stretches.	Kubecka *et al.* (1997)
River Ardena, Portugal (Q_{mean}: drought season 0.53 m^3/s, rainy season 1.77 m^3/s)	Unknown	• Loss of wetted area and accumulation of fine sediment and organic matter in depleted stretch.	• Invertebrate density and richness reduced in depleted stretch and downstream reaches. • Replacement of lotic, sensitive taxa with lentic, less sensitive taxa.	Jesus *et al.* (2004)
20 schemes on River Gave de Pau, France (Q_{mean} 90 m^3/s)	Unknown	• Absent from study	• 17% barriers posed major obstacles to fish migration, whereas 33% caused partial delay.	Larinier (2008)
18 schemes on rivers (median stream order 4) across central and northern Portugal	0.3–8.7 MW. Turbine capacity 2–30 m^3/s	• Absent from study	• Smaller fish found upstream of scheme particularly when fish pass unsuitable for population.	Santos *et al.* (2006)
River Hoz Seca, central Spain (Q_{mean}: summer 0.4 m^3/s, winter 2 m^3/s)	700 kW	• Successive increase and decrease of discharge and water level downstream of scheme.	• Invertebrate density and biomass increased downstream of scheme. • Brown trout density and biomass were significantly reduced and the population became dominated by older fish.	Almodovar & Nicola (1999)

These studies were identified from an extensive literature search using Web of Science, Thomson Reuters Corporation, New York, USA.

sediment (Skalak *et al*. 2009; Csiki & Rhoads 2010), particulate organic matter (Pohlon *et al*. 2007), nutrients (Stanley & Doyle 2002), aquatic species (Benstead *et al*. 1999; O'Connor *et al*. 2006) and plant propagules (Jansson *et al*. 2000). The upstream and downstream movement of fish (e.g. salmonids, lamprey, eel, some cyprinids) is also affected, preventing access to spawning or feeding grounds and threatening life-cycle completion, whether species migrate between sea and river, or just within the river (Lucas & Frear 1997; Aarestrup & Koed 2003; Lucas *et al*. 2009; Gauld *et al*. 2013). These latter effects may be limited for high-head schemes, where existing topographical features may already create natural barriers to fish migration, but may be more significant in low-head schemes without any mitigation (see *Potential for mitigation*).

Weirs also alter the nature of the physical habitat. The raised water levels upstream of weirs reduce flow variability, velocity and turbulence and induce fine sediment deposition (Csiki & Rhoads 2010; Mueller *et al*. 2011) creating a lentic environment ('weir pond'; Fig. 1) that can extend for several kilometres (Walter & Merritts 2008). These environments exhibit lower biodiversity and distinctly different populations of benthic algae and macrophytes (Mueller *et al*. 2011), macroinvertebrates (Arle 2005; Mueller *et al*. 2011), riparian vegetation (Jansson *et al*. 2000; Greet *et al*. 2011) and fish (Miranda *et al*. 2005; Mueller *et al*. 2011) relative to unimpounded reaches. In contrast, the higher velocity, more turbulent and sediment-deprived flows downstream of weirs ('weir pool' habitat) erode bed sediment, creating scour holes, bank undercutting and downstream bar formation (Skalak *et al*. 2009; Csiki & Rhoads 2010), although this is fairly localised around the base of the weir (Shaw 2012). Evidence of ecological impacts downstream of weirs is sparse, but it has been suggested that the increase in habitat diversity may be beneficial, providing spawning areas for fish and key habitats for macrophytes and invertebrates, particularly in large, modified rivers (Shaw 2012; EA 2013c).

Studies on the impact of barriers specific to hydropower schemes have largely focussed on the passage of migratory fish. While most demonstrate that barriers hinder fish migration (see Kubecka *et al*. 1997; Larinier 2008 in Table 1), Santos *et al*. (2006) did not find significant differences in fish species, abundance or diversity upstream or downstream of hydropower schemes. Migration impediment may result from the physical barrier, the increased presence of unsuitable habitat from alteration of physical conditions, or from lethal and sublethal passage through turbines or poorly designed 'protective' screens. IEA (2000) and Robson *et al*. (2011) provide detailed reviews of the potential direct mechanical impacts of ROR schemes on fish.

Impacts of water abstraction/flow depletion

The diversion of flow through a hydropower scheme creates a stretch of river (from the point of abstraction to return) that is depleted of water while the scheme is operating, but has natural flows when it is not (Fig. 1). This alteration of flow regime can alter the physical habitat, with consequences for organisms and ecosystem functions (Poff *et al*. 1997; Biggs *et al*. 2005) and habitat connectivity (Ward 1989; Tockner *et al*. 2000).

Impacts on riverine habitat

Depleting flows for hydropower has been found to reduce lotic habitat in depleted stretches (see Anderson *et al*. 2006; Kubecka *et al*. 1997; Ovidio *et al*. 2008; Jesus *et al*. 2004 in Table 1). Other studies on flow depletion (not specifically for hydropower) have also shown changes in habitat availability and water chemistry (McKay & King 2006) and reduced in-stream habitat complexity (McIntosh *et al*. 2002; Riley *et al*. 2009). That said, habitat heterogeneity was unaffected by a high-head hydropower scheme in the UK (Copeman 1997).

The reduction of habitat confines biota and may increase competition for food and space (McIntosh *et al*. 2002; Riley *et al*. 2009), potentially increasing dispersal to more suitable habitats downstream (Davey *et al*. 2006). Stretches affected by flow depletion have been found to exhibit altered riparian vegetation (often with reduced proportions of riparian species) (Elderd 2003; Greet *et al*. 2011), macroinvertebrate (McIntosh *et al*. 2002; McKay & King 2006) and fish (Richter *et al*. 2003; Benejam *et al*. 2010) communities (see Poff & Zimmerman 2010, for a review). Studies on hydropower-depleted stretches show contrasting results: Jesus *et al*. (2004) found reductions in Ephemeroptera, Plecoptera and Trichoptera, whereas other studies show negligible impacts on invertebrates (Copeman 1997; Kubecka *et al*. 1997). Reported impacts on fish in hydropower-depleted reaches include reductions in biomass (Kubecka *et al*. 1997; Habit *et al*. 2007), changes in species composition (Anderson *et al*. 2006) and shifts in population structure (Ovidio *et al*. 2008), although other studies have not detected significant impacts (Robson 2013). However, these findings must be considered in the context of the schemes (see Table 1); many of the low-head studies were on much larger schemes than those found within Europe, or were not subject to the same legislative requirements for flow retention. Notably, there are no studies on in-weir schemes, despite their recent expansion and potential impacts on weir pool habitat from flow depletion/hydraulic alteration.

Impacts on connectivity

The disruption of longitudinal connectivity from in-channel barriers is exacerbated by reduced flows passing over the barrier (Aarestrup & Koed 2003; Lucas *et al*. 2009; Gauld *et al*. 2013). Diadromous and potadromous species follow the major flow when migrating downstream and diversion of this into hydropower schemes could encourage species to enter schemes, resulting in injury or mortality (Larinier 2008; Svendsen *et al*. 2010). Where screening prevents entry, the lack of a suitable bypass may still impede migration as the reduced flow over the weir crest may discourage downstream passage (Gauld *et al*. 2013). In-weir schemes with slower rotating devices (e.g. Archimedean screws) may have less impact, as many fish can pass through such schemes unharmed (Kibel *et al*. 2009; Bracken & Lucas 2013). Such schemes are typically only coarsely screened.

Diversion schemes may also hinder upstream migration through the creation of unsuitable habitat in depleted stretches and reduction in the required hydraulic conditions for weir passage (Lucas *et al*. 2009). Migratory species may be attracted to hydropower discharges where these provide a more powerful stimulus than depleted channel flow, which could further reduce migration success (Arnekleiv & Kraabol 1996; Williams *et al*. 2012). This exacerbation of a barrier's 'bottle neck' effect may increase predation risk and thus population isolation and loss (Benstead *et al*. 1999; De Leaniz 2008; Lucas *et al*. 2009).

Existing studies focus on the impact of individual schemes, although future ROR hydropower is likely to involve multiple schemes installed along a river. Where impacts of individual schemes are modest, the cumulative effects of multiple schemes, particularly on species migration, may be much greater. Understanding these cumulative effects and their interaction with other anthropogenic stressors is an important and under-researched issue (Larinier 2008; Robson 2013), although see Cada & Hunsaker (1990).

Potential for mitigation

A number of measures are available to attempt to mitigate the impacts from ROR hydropower schemes. The most common technique for mitigating against flow depletion involves setting minimum flow requirements for the depleted stretch. For example, UK policy (driven by EU WFD requirements) requires maintenance of a 'hands off' flow in depleted stretches, where schemes only operate when flows exceed a particular threshold. This varies with scheme location and type, but is typically between Q85 and Q95 (EA 2013b). There is no agreement on what represents a suitable value, with notable variation within Europe (ESHA 2008) and studies questioning the adequacy of such requirements for fish (Robson 2013). Alternative approaches include 'flow splitting', where flow abstraction (above 'hands off') is proportionally split between the channel and the scheme, promoting a more natural flow regime, or ceasing scheme operation during key stages of fish life cycle. The effectiveness of these alternative approaches is unknown (EA 2013g).

Current mitigation of the barrier effects of weirs has concentrated on fish pass installation. The UK focus has predominantly been on upstream passage, with varying designs from rock ramps and natural bypass channels to highly engineered structures, for example Denil, Larinier, weir and pool fishways (Larinier 2008). Fish passes for downstream movement have received greater emphasis in North America and Europe, although uptake is increasing in the UK (Roscoe & Hinch 2010; EA 2013a). Downstream passes are typically highly engineered structures, for example spill ways or screened surface bypass collectors, but natural bypass channels may also be functional (IEA 2000). Low flow notches on the weir crest may also reduce the downstream barrier effect (EA 2013a; Gauld *et al*. 2013).

The effectiveness of both upstream and downstream fish passes is understudied (Roscoe & Hinch 2010), with existing research focussing on efficiency for salmonid passage (Roscoe & Hinch 2010; Noonan *et al*. 2012). The effectiveness of a fish pass is dependent on appropriate situation, and adequate flow provision, and suitable design (slope, length, water velocity) for target species (Noonan *et al*. 2012). Where hydropower schemes create notable depleted reaches and screening of the tail race is impractical (e.g. low-head diversion schemes), multiple fish passage facilities may be required. The utility of fish passes for improving connectivity for other species, for example invertebrates, remains under-researched.

Fish passage through schemes can be mitigated through screening to prevent entry. Screening requirements depend on turbine type and the native fish population (see Turnpenny *et al*. 1998 and EA 2013e for further detail). Although regulations on fish screening are strictly enforced within the UK (EA 2013e), legislation and enforcement varies within Europe (Turnpenny *et al*. 1998), and outside of Europe, screens do not seem to be routinely installed.

Hydropower schemes may also have potential to improve ecological function, particularly where schemes are retrofitted to existing structures and effects from the in-channel barrier and flow depletion already occur. For instance, ROR schemes often provide impetus for the installation of fish passes on structures that may previously have posed serious migratory barriers (EA 2013a). Additionally, hydropower installations may increase loticity of weir ponds by drawing a large flow from a single point, rather than over a long weir crest (D. Anderson, University of Sheffield, personal observation). This may improve downstream movement of sediment and biota, although further research is required. A study of ROR hydropower feasibility in the UK suggested that 49% of potential hydropower sites could provide both energy and ecological improvement ('win-win') (EA 2010). Currently, there is no requirement or subsidy in the UK for these win-win situations, although they are favoured (EA 2013d, 2013f).

Future research needs

This review has highlighted the general lack of peer-reviewed studies on the physical and ecological impacts of ROR hydropower schemes. Current understanding is largely based on comparisons with large, storage hydropower schemes or water abstraction studies (e.g. Robson *et al*. 2011). Directed research has focussed on fish and suggests adverse impacts, but this must be interpreted with caution; findings are variable and have limited general application because of the size of studied schemes, waterbody condition, the native ecological community and the absence of any mitigation measures. Research in this field is constrained by the absence of long-term data and the capacity to evaluate temporal changes following scheme installation; current studies mainly rely on spatial comparison with 'control' reaches, which limits the conclusions that can be drawn (Robson 2013). Retrofitting hydropower to existing structures presents a further challenge in isolating hydropower impacts from those associated with existing modifications. It is also important to separate impacts associated specifically with hydropower from impacts due to poor implementation, operation or mitigation.

Attempts have been made to address this deficit. Scheme proposals in the UK, for example, increasingly require baseline hydromorphological and ecological data and modelling of the likely hydraulic impact on weir pool habitats to ensure schemes do not compromise WFD targets (EA 2013c, 2013d, 2013f). However, this is not always required and data are often unavailable or not collected appropriately (e.g. with sufficient replication and/or spatial resolution) for use in longer-term scientific study. The extent of temporal ecological monitoring varies between schemes; long-term ecological monitoring is generally only performed at 'environmentally sensitive' sites, whereas ensuring sufficient 'hands off' or fish pass flows constitutes the only form of monitoring at most sites (EA 2013d). More detailed and consistent monitoring is required, although funding this is a constraint. Funding is currently the responsibility of the developer in the UK and increased requirements could threaten the economic viability of schemes and thus hinder achieving renewable energy targets. Increasing monitoring requirements would therefore require further government subsidy. Monitoring only partially addresses the issue however and must be complemented by scientific research studies to gain a more detailed insight into the impacts of schemes and the effectiveness of mitigation measures. Planning, obtaining funding and executing a systematic before–after comparison study on a proposed scheme can be challenging, however, because of the unpredictable nature of scheme implementation.

Current evidence does suggest that ROR hydropower can alter habitat availability and the structure of biological communities locally and has the potential to impact physical and ecological processes (such as sediment transport and fish migration) at larger spatial and temporal scales, particularly if multiple schemes occur on a single river. However, there is insufficient evidence to evaluate the ecological significance of these effects. The current expansion of hydropower presents an urgent need to understand these impacts, particularly in light of legislative requirements to improve the ecological condition of waterbodies (e.g. EU WFD).

Conclusions

We identify the following priority areas and approaches for further research:

- To get a true understanding of the impacts of ROR schemes, we must attempt to move away from the opportunistic *post hoc* investigation of schemes and instead implement long-term, large-scale, 'experimental' before–after control–impact studies. This requires a catchment-level approach, so that comparable control sites can be retained and monitored alongside hydropower sites. Implementing such studies would require significant financial investment from national governments or transnational sources (e.g. EU or United Nations) to be feasible.
- In the absence of available funding, a general increase in the quantity and usability of baseline hydromorphological and ecological data would still enable robust before–after studies. Collecting these data requires cooperation between researchers, river managers and scheme developers to enable data to be collected between scheme proposal and installation. Flexibility in research funding to accommodate timeframes and uncertainties of scheme proposal would facilitate this. Ecological studies must broaden from impacts on diadromous fish to consider all mobile and sedentary fish species, as well as key benthic invertebrates, macrophytes and riparian vegetation communities. In the absence of baseline data, a more comprehensive, interdisciplinary understanding of ecosystem processes is needed to predict likely impacts from hydropower. For example, an improved understanding of the relationships between biota of different trophic levels and channel hydraulics would allow prediction of response to disturbances from hydropower schemes at different magnitudes and over spatial and temporal scales. Numerical modelling of the hydraulic impacts of water abstraction and flow depletion would complement field data collection and make impact studies more comprehensive and cost-effective (and thus more feasible).
- We highlight that potential impacts will vary notably among scheme type (high-head, low-head diversion and low-head in-weir). Thus, impact studies need appropriate focus and should avoid between-scheme comparisons. Studies on

schemes with significant depleted reaches should focus on identifying the impact of the hydropower-altered flow regime through continuous temporal discharge monitoring and the use of numerical modelling to identify impacts on hydraulics and habitat availability. Studies exploring differences in the physical habitat and ecology of depleted reaches relative to the natural variability of the river would also be of great value. The major impact in low-head, in-weir schemes is the hydraulic alteration of weir pool and weir pond habitats and research should focus on characterising this through a combination of numerical modelling and comparative on/off field studies (where possible). These schemes are particularly understudied, yet increasingly widely implemented in the UK and Europe, so further research is essential. In all scheme types, fish-tagging studies that investigate impacts on migration, dispersal and mitigation measure effectiveness are needed.

• Increased understanding of the cumulative impact of multiple schemes on a watercourse is needed. Such studies will need to draw on known biotic capabilities (e.g. fish swimming capability) and habitat preferences, as well as utilising existing knowledge of impacts from comparable water abstraction and barrier studies and use a combination of catchment scale hydrological, habitat and ecological modelling tools to suggest the impacts of expanded development at broader spatial scales.

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