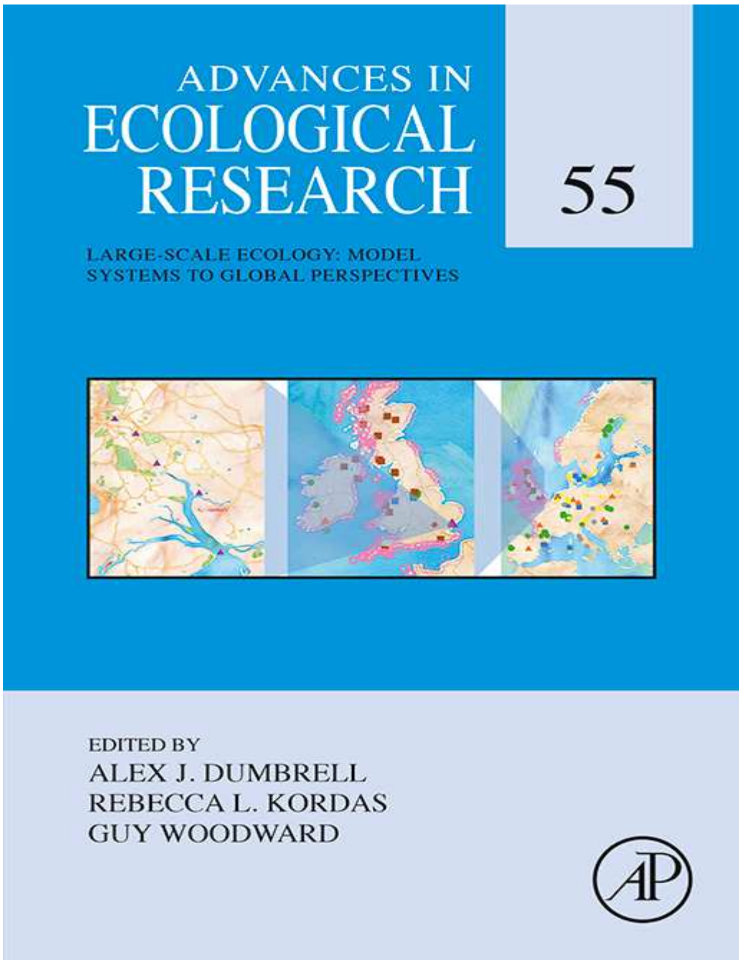

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CHAPTER ELEVEN

Effective River Restoration in the 21st Century: From Trial and Error to Novel Evidence-Based Approaches

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Abstract

This paper is a comprehensive and updated overview of river restoration and covers all relevant aspects from drivers of restoration, linkages between hydromorphology and biota, the current restoration paradigm, effects of restorations to future directions and ways forward in the way we conduct river restoration. A large part of this paper is based on the outcomes of the REFORM (REstoring rivers FOR effective catchment Management, http://reformrivers.eu/) project that was funded by EU's 7th Framework Programme (2011–15). REFORM included the most comprehensive comparison, to date, of existing river restorations across Europe and their effect on biota, both in relation to preintervention state and project size in terms of river length restored. The REFORM project outcomes are supplemented by an extensive literature review and two case studies to illustrate key points. We conclude that river restorations conducted up until now have had highly variable effects with, on balance, more positives than negatives. The largest positive effects have interestingly been in terrestrial and semiaquatic organism groups, in widening projects, while positive effects on truly aquatic organisms groups are only seen when in-stream measures are applied. The positive responses of biota are primarily seen as increased abundance of organisms with very little indication that overall biodiversity has increased: specific traits rather than mere species number or total abundance have benefited from restoration interventions. This modest success rate can partly be attributed to the fact that the catchment filter is largely ignored; large-scale pressures related to catchment land use or the lack of source populations for the recolonisation of the restored habitats are inadequately considered. The key reason for this shortfall is a lack of clear objective setting and planning processes. Furthermore, we suggest that there has been a focus on form rather than processes and functioning in river restoration, which has truncated the evolution of geomorphic features and any dynamic interaction with biota. Finally, monitoring of restoration outcomes is still rare and often uses inadequate statistical designs and inappropriate biological methods which hamper our ability to detect change.

1. INTRODUCTION

1.1 A Brief Introduction to River Restoration

Restoration is a human intervention aimed at improving conditions of an ecosystem to a former, predisturbance state. Inherently, the term 'restoration' elicits confusion as projects are very rarely aimed at restoring ecosystems back to exactly the way they used to be. This is certainly true for rivers, which are often nested within catchments where larger scale impacts such as land use are the rule rather than the exception (Friberg, 2014). In many ways, the definition by the Society of Ecological Restoration (SER, 2004) from 2004 stating that '*Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed*' was important in focusing a discussion that had become more about semantics than science. Importantly, and in the context of the present paper, the definition is to the point when dealing with the restoration of rivers and fluvial systems, where the main focus has been primarily on smaller reach-scale interventions to improve, longitudinal connectivity, channel planform (the configuration of the river channel in plan view, e.g. meandering) and local habitat conditions (Bernhardt et al., 2005; Feld et al., 2011). Removing dams, weirs and other obstacles for migration of primarily diadromous fish have been, and still are, a very common type of restoration intervention (Kail et al., 2015). Change of planform, in particular from a straight channel to a meandering course, has become popular in river restoration (Feld et al., 2011; Kail et al., 2015); unfortunately, this has been indiscriminate, including instating meanders where they are not historically a naturally occurring geomorphological feature (Walter and Merritts, 2008). Concepts such as physical habitats and mesohabitats are frequently used in river restoration projects as synonyms for improving substrate conditions and increase variability in water velocities for the expected benefit of aquatic species (Friberg, 2010).

River restoration has been high on the global environmental agenda for more than 3 decades with a large number of projects undertaken primarily in Europe and the United States (Feld et al., 2011; Ormerod, 2004; Palmer et al., 2005). Investments have been significant and river restoration has received wide recognition in the public, reflected by the engagement of both stakeholders and scientists. Many previously degraded rivers look different today compared with the recent past and this change in appearance is a very visual outcome of restoration. However, the history of river restoration is also flawed by the lack of a systematic approach in project development and infrequent evaluation of ecosystem responses (Bash and Ryan, 2002; Bernhardt et al., 2005). In the past decade, there has been some improvement with an increasing number of scientific papers that review or report results on effects of river restoration (e.g. Kail et al., 2015; Palmer et al., 2010; Roni et al., 2008; Whiteway et al., 2010).

The capacity of practitioners to implement and achieve ecosystem restoration is often limited by opportunity, time or economic constraints (Borgström et al., 2016). So despite considerable investment, river restoration still depends on trial and error as there is a lack of comprehensive and well-formulated planning, implementation and appraisal techniques. Several guidance documents have been produced over the last few decades to either assist with planning and techniques (Cowx and Welcomme, 1998; FISRWG, 1998; Roni and Beechie, 2013; RRC, 2011; Ward et al., 1994) or provide overviews of key concepts and principles (Brierley and Fryirs, 2008; Clewell and Aronson, 2008). Collectively these publications cover many of the tools, techniques and concepts needed for restoration activities, but not the planning and integration of restoration processes from initial assessment to monitoring of results. With an increased emphasis on restoration has come the need for new techniques and guidance for assessing stream and catchment condition that has the necessary sensitivity to detect postrestoration changes (Rumps et al., 2007). Another weakness of the current river restoration paradigm is that societal needs, such as energy demands, freshwater and food supply, transportation networks, flood protection, etc., have rarely been considered and other potential socioeconomic benefits, e.g. increasing value of private properties, are seldom fully explored, understood or achieved (Ayres et al., 2014; Wortley et al., 2013).

1.2 The Need for Restoration

Ecosystems worldwide face large-scale challenges such as population growth, climate change, land degradation and habitat loss (Foley et al., 2005; Halpern et al., 2008; Parmesan and Yohe, 2003; Sanderson et al., 2002). Freshwater ecosystems, and in particular rivers, have a long history of human pressures and they have been impacted by a number of very deteriorating types of stress relating primarily to sewage influx, land-use intensification and physical degradation. As a result of this sum of stressors, the ecological condition of rivers is still highly impaired and globally these ecosystems rank among those that have seen the greatest loss of biodiversity

(Sala et al., 2000; Vörösmarty et al., 2010).The negative impacts of such pressures on biodiversity and ecosystem services, e.g. the provision of freshwater, are recognised worldwide and have led to a set of international targets including the UN Aichi Biodiversity Target 15 and EU 2020 Biodiversity Strategy Target 2 of restoring at least 15% of degraded ecosystems by 2020 (Convention on Biological Diversity, 2010; European Commission, 2011).

The scope for freshwater restoration is vast with more than 50% of Europe's freshwaters not meeting their environmental quality objectives of good ecological status (GES) or potential (Solheim et al., 2012), despite a costly and largely successful, effort to improve in particular water quality, e.g. lowering concentrations of easy degradable organic matter from sewage (BOD_5) that depletes oxygen in the water. The wide range of services (Millennium Ecosystem Assessment, 2005) that can be delivered by a functioning ecosystem is the target of a multitude of policies and legislations, e.g. the Paris Agreement on Climate Change, the EU Renewable Energy Sources Directive, the UNEP Green Economy Initiative and the US Clean Water Act. Consequently, the focus of river restoration is now predominantly on biodiversity and the delivery of ecosystem services such as carbon sequestration, flood protection and provision of freshwater (Palmer et al., 2014). However, restoring previously degraded physical habitat features in rivers and floodplains affects the interests of multiple stakeholders (Kondolf and Yang, 2008; TEEB, 2011) and implementation of more recent environmental quality targets, such as those specified by the Water Framework Directive (WFD) of the EU or the Aichi Biodiversity Targets, has encountered resistance, slowing down implementation and yielding mixed success (Hart et al., 2012; Jähnig et al., 2011).

The assessment of the first river basin management plans (RBMPs), as part of implementing the EU WFD (see Box 1) indicated that 40% of European rivers are affected by changed hydrological regimes and degraded channel morphology caused predominantly by hydropower, navigation, agriculture, flood protection and urban development. The aim of WFD is that all water bodies should fulfil a requirement of GES by 2027 at the latest. As a consequence, there is increasing emphasis in Europe on river restoration driven by demands of the WFD, where restoring hydrology and morphology, as part of programmes of measures (PoMs), is an important component in achieving GES. Implementation of PoMs requires substantial investment in these measures, but there still remains a great need to better understand and predict the costs and benefits of future river restoration. Furthermore, ecological response to river restoration is complex and poorly understood.

BOX 1 Important Policy Drivers for River Restoration

There are a number of European Directives to support the ecological health of rivers such as the Water Framework Directive (WFD (2000/60/EC)), Habitats Directive (HD (92/43/EEC)) and Groundwater Directive (GWD (2006/118/EC)), in addition to global initiatives such as Agenda 21 of the Rio Convention and the Convention of Biological Diversity. These have driven the management of inland waters towards rehabilitation of rivers and lakes to improve the aquatic environment for biodiversity and allow for sustainable exploitation of the resources (Hobbs et al., 2011; Pasternack, 2008). Consequently, nature conservation, and in particular river restoration, is increasingly considered as part of a much wider framework of environmental policy and practise (Arlinghaus et al., 2002). In Europe, the main policy driver of restoration is without a doubt WFD that was implemented in 2000, and to understand parts of the present paper it is necessary to introduce some key concepts of the Directive. One of the first steps of implementing the Directive was to identify river basins that could be overall management units and some of these traverse national frontiers. River Basin Management Plans (RBMPs) have to be established and updated every 6 years in three plan periods starting 2009 and ending in 2027. RBMPs are detailed accounts of how objectives set for the river basins (ecological status, quantitative status, chemical status and protected area objectives) are to be reached within the timescale required. The plan will include river basin characteristics, a review of the impact of human activity on the status of waters in the basin, estimation of the effect of existing legislation and the remaining 'gap' to meeting these objectives; as well as a set of measures designed to bridge the gap.

1.3 Drivers of River Restoration

Degradation and loss of physical complexity in river ecosystems have been massive in most parts of Europe through, for instance, channellisation, dredging, wood removal, etc. (Friberg, 2010). In addition, siltation with fine sediments is a major problem in many streams, especially in agricultural catchments (Glendell et al., 2014). The importance of habitat heterogeneity for biota is indisputable but surprisingly few studies have documented clear impacts of habitat degradation (Friberg, 2014). In the following, the collective term *hydromorphology* is used for the physical environment as it is used by the WFD, signifying how important physical features are considered in determining the ecological status of freshwaters. The hydromorphological quality of a river in 'high' status class is defined as follows: '*Channel patterns, width and depth variations, flow velocities, substrate conditions and both structure and* *condition of the riparian zones correspond totally or nearly totally to undisturbed conditions*'. With regard to the other status classes, hydromorphology is considered a supporting element and there are no normative definitions but can help in explaining why biological quality elements (BQEs) are not achieving GES. It is evident from the RBMPs undertaken thus far that degraded hydromorphology is one the most extensive impacts on river ecosystems in Europe today. Centuries of modification by man to ensure drainage, flood protection, navigation and hydropower have completely altered habitat area, channel form and processes in rivers and floodplains almost everywhere. While the effects of stressors such as low oxygen levels on the river biota are well documented and have been instrumental in reducing sewage loads, specific methods to assess the impact of degraded hydromorphology on ecological status are relatively uncommon. Provisioning and regulating services from development sectors (pressures: Table 1; Fig. 1), such as water resource management, flood protection, inland navigation, hydropower and agriculture, have led to the replacement of naturally occurring and functioning systems with highly modified and human-engineered systems.

Fig. 1, furthermore, exemplifies how complex interactions between the main drivers of hydromorphological degradation, in this case damming of rivers and creating large reservoirs, and a range of associated environmental factors will substantially change living conditions for the biota across scales. Consequently, damming a river will have profound impacts beyond impairing longitudinal connectivity that directly affects anadromous fish species. This complexity in response to hydromorphological degradation signifies the importance of appropriate restoration strategies.

1.4 A Short Introduction to the REFORM Project and Scope of this Paper

A large part of this paper is based on the outcomes of the REFORM (REstoring rivers FOR effective catchment Management, http://reformrivers.eu/) project that was funded by EU's 7 Framework Programme (2011–15) and brought together 26 research institutes and applied partners from 15 European countries (Fig. 2).

The overall aim were to generate tools for cost-effective restoration of river ecosystems, and for improved monitoring of the biological effects of physical change by investigating natural, degradation and restoration processes in a wide range of river types across Europe. In relation to this chapter, the following specific aims are relevant:

Table 1 Identification of Main Pressures on Rivers and their Categorisation in Specific Types of Pressures

Main Pressure	Specific Types of Pressures
Water abstraction	Groundwater abstraction Surface water abstraction
Flow regulations	Discharge diversions and returns Hydrological regime modification: timing or quantity Hydropeaking Interbasin flow transfers Reservoir flushing Sediment discharge from dredging
River fragmentation	Artificial barriers downstream from the site Artificial barriers upstream from the site Colinear connected reservoir Large dams and reservoirs
Morphological alterations	Alteration of in-stream habitat Alteration of riparian vegetation Channellisation/cross-section alteration Embankments, levees or dikes Impoundment Loss of vertical connectivity Sand and gravel extraction Sediment input
Water quality	Diffuse source pollution Point source pollution

The majority of pressures relates to physical changes of rivers, i.e. hydrological and geomorphological alterations often denoted 'hydromorphology' in accordance with the EU WFD.
After Garcia de Jalón, D., Alonso, C., et al., 2013. Review on pressure effects on hydromorphological variables and ecologically relevant processes. REFORM Deliverable D1.2, Report to the European Union (available at http://reformrivers.eu/).

- Will river biota show a sufficiently strong response to hydromorphological change, either in terms of degradation or restoration that the effects can be singled out from other stressors acting on ecosystem as well as effects of biotic interactions and dispersal mechanisms?
- Which types of river restorations are the most frequently used in Europe and how are these in accordance with the main challenges regarding hydromorphological degradation?
- What are the effects of river restoration on the various components of the river biota and how do spatial and temporal scales influence the outcomes?

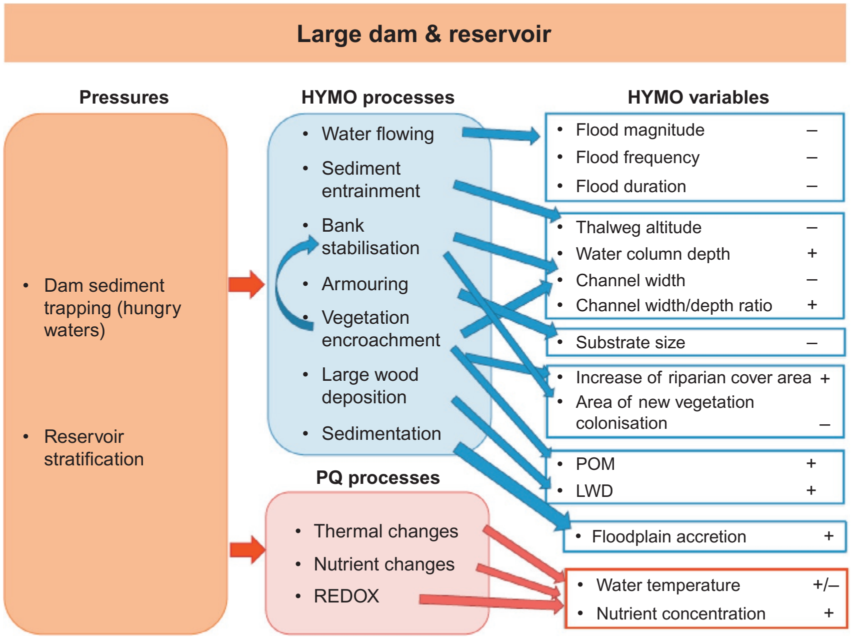
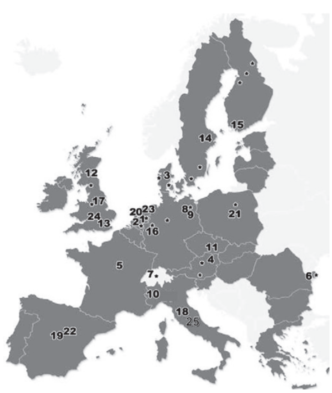


Fig. 1 The link between hydromorphological pressures, the processes they impair and their effect on key variables that is important for river biota here exemplified with large dams and reservoirs as the overall pressure. It is evident that this pressure has a multitude of effects on habitat conditions severely degrading living conditions for a number of species (Garcia de Jalón et al., 2013). *HYMO*: Hydromorphological processes; *PQ*: physical–chemical processes; *LWD*: large woody debris; *POM*: particulate organic matter.



No	Name	Short name	Country
1	Stichting Deltares	Deltares	Netherlands
2	Stichting Dienst Landbouwkundig Onderzoek	Alterra	Netherlands
3	Aarhus University	AU-NERI	Denmark
4	Universitaet fuer Bodenkultur Wien	BOKU	Austria
5	Institut National de Recherche en Sciences et des Technologies pour l'Environnement et l'Agriculture	IRSTEA	France
6	Institutul National de Cercetare-Dezvoltare Delta Dunarii	DDNI	Romania
7	Swiss Federal Institute of Aquatic Science and Technology	EAWAG	Switzerland
8	Ecologic Institut Gemeinnützige Gmbh	Ecologic	Germany
9	Forschungsverbund Berlin E.V.	FVB.IGB	Germany
10	Joint Research Centre-European Commission	JRC	Belgium
11	Masaryk University	MU	Czech Republic
12	Natural Environment Research Council - Centre for Ecology and Hydrology	NERC	United Kingdom
13	Queen Mary University of London	QMUL	United Kingdom
14	Swedish University of Agricultural Sciences	SLU	Sweden
15	Finnish Environment Institute	SYKE	Finland
16	Universitaet Duisburg-Essen	UDE	Germany
17	University of Hull	UHULL	United Kingdom
18	Universita Degli Studi Di Firenze	UNIFI	Italy
19	Universidad Politecnica de Madrid	UPM	Spain
20	Stichting VU-VUmc	VU-Vumc	Netherlands
21	Warsaw University of Life Sciences	WULS	Poland
22	Centro de Estudios y Experimentacion de Obras Publicas	CEDEX	Spain
23	Dienst Landelijk Gebied	DLG	Netherlands
24	Environment Agency	EA	United Kingdom
25	Istituto Superiore per la Protezione e la Ricerca Ambientale	ISPRA	Italy
26	Norsk Institutt for Vannforskning	NIVA	Norway

Fig. 2 The REFROM project comprised of 26 partners from 15 European countries covering all major landscapes and river types.

- How can river restoration become more efficient in the future in delivering expected outcomes and secure that environmental objectives are achieved as well as stopping the decline in biodiversity?

These aims are addressed by a combination of reviewing the literature and undertaking new analysis on existing datasets combined with field sampling. REFORM included the most comprehensive comparison to date of existing river restorations across Europe and their effect on the biota, both in relation to preintervention state and project size in terms of river length restored (Fig. 3).

We provide a comprehensive and fully updated status for restoration activities in Europe that hopefully can help to steer future developments in the right direction. In the text we refer to the 'REFORM project' whenever we use primary results if these are not already published. Full details regarding methods and also additional results that we cannot cover here can be found on the project home page.

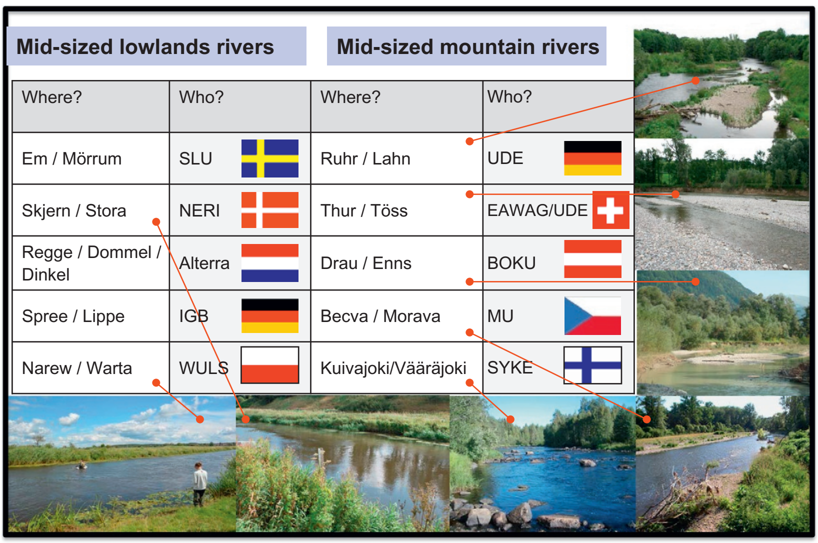


Fig. 3 Ten pairs of river restoration projects differing in size (predominantly river length) were investigated as part of REFORM across Europe. The restoration effect of the large and smaller project size was investigated in a similar length river section and quantified by comparing each of these to a nearby nonrestored section (space-for-time substitution). We sampled the following response variables: habitat composition in the river and its floodplain, three aquatic organism groups (macrophytes, benthic invertebrates and fish), two floodplain-inhabiting organism groups (floodplain vegetation and ground beetles), as well as food web composition and aquatic land interactions as reflected by stable isotopes (Hering et al., 2015).

2. RESPONSES OF RIVER BIOTA TO HYDROLOGY AND PHYSICAL HABITATS

2.1 Can We Expect the Biota to Respond to River Restoration?

Natural rivers depend on catchment-scale structural controls, reach-scale channel pattern differences and microscale variations in channel bed forms, all of which vary over different time scales (Friberg, 2014). In this hierarchical organisation, structure and processes occurring on small spatial and temporal scales are nested within increasingly larger scales, from microhabitat to catchment (Fig. 4).

Naturally, therefore, the effects of restoration will be scale dependent and linked to the spatial and temporal heterogeneity provided by natural stream reaches, which creates a range of biotopes for the biota and is the scale where impact is assessed (Frissell et al., 1986; Wolter et al., 2016). Moreover, lotic ecosystems will be impacted at a range of scales depending on the type of pressure, with larger scale impacts having negative effects on lower levels

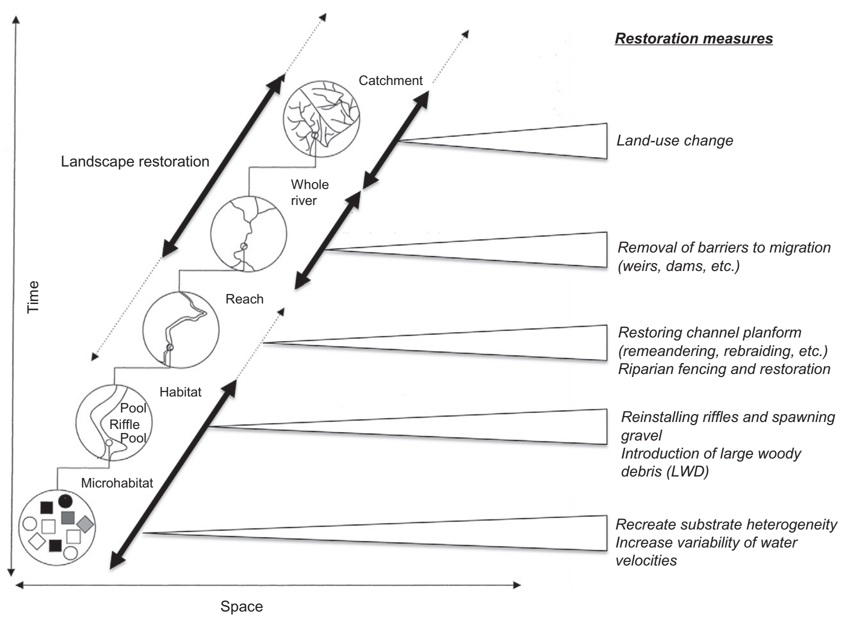


Fig. 4 Effects of restoration is dependent of both spatial and temporal scales. Furthermore, the type of restoration measure is scale dependent.

of organisation. A key challenge to the restoration of rivers is the assumption ‘if you build they will come …’ which has been the core paradigm in most projects (Palmer et al., 1997). Local physical habitat conditions are without doubt important filters for the biota and species have evolved life histories in response to coarse substrates originating from riverine sediment sorting, supporting the assumption that there is a clear causative link between habitats provided at local scales and biological communities present (Fig. 5).

In effect, other local-scale controls such as biotic interactions and dispersal at larger scales must be of insignificant importance for this assumption to hold and this is clearly not valid in all cases. Local environmental conditions are not the sole determinant of macroinvertebrate or fish community composition as both local biotic interactions (Woodward, 2009) and larger dispersal mechanisms (Heino, 2013; Radinger and Wolter, 2015) play an important structuring role.

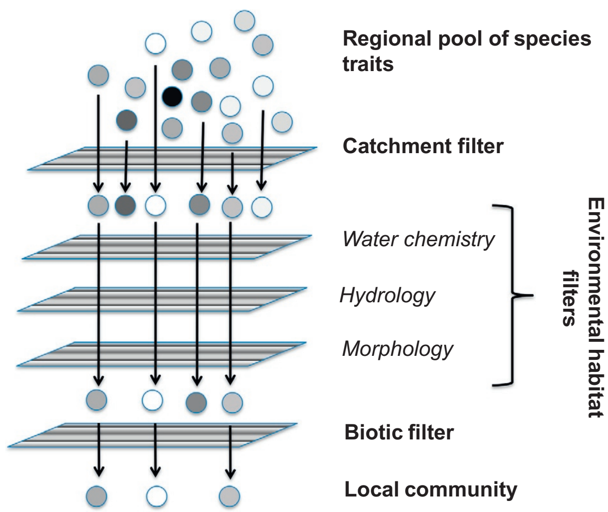


Fig. 5 Local community composition of species traits is dependent on the filtering of the regional pool of species traits by (a) catchment filters; (b) environmental habitat filters and (c) local biotic filters. In the context of river restoration, the focus has been primarily to modify the hydrological and morphology components of the environmental habitat filters. However, it is evident that filtering at the scale of catchments can reduce the potential of recovery even if local habitat conditions are improved (Stoll et al., 2016). Likewise, water chemistry will also be an important filter and poor quality will impair possibilities of recovery if not considered (Violin et al., 2011). The biotic filter is largely ignored in restoration projects although it is important in determining local community composition.

Restoration measures potentially influence river food webs at each trophic level with implications up through the web in terms of, for example, energy transfer (Fig. 6). For instance, increased retention via the restoration of more complex bed forms will increase retention of allochthonous organic matter, which can further be increased by rehabilitating riparian zones so they include a large proportion of woody vegetation. This will increase the detrital food base and increase detritivore biomass, which are important food items for higher trophic levels. In wider rivers with less riparian shading in-stream macrophytes have the ability to increase physical heterogeneity and improve habitat conditions for top predators such as fish. Overall, restoration has the potential to increase the resource base of river ecosystems with knock-on effects throughout the food web. However, while this is at the core of biomanipulation in lakes (Søndergaard et al., 2007), restoration projects that consider the effects at the scale of the food web do not exist for rivers.

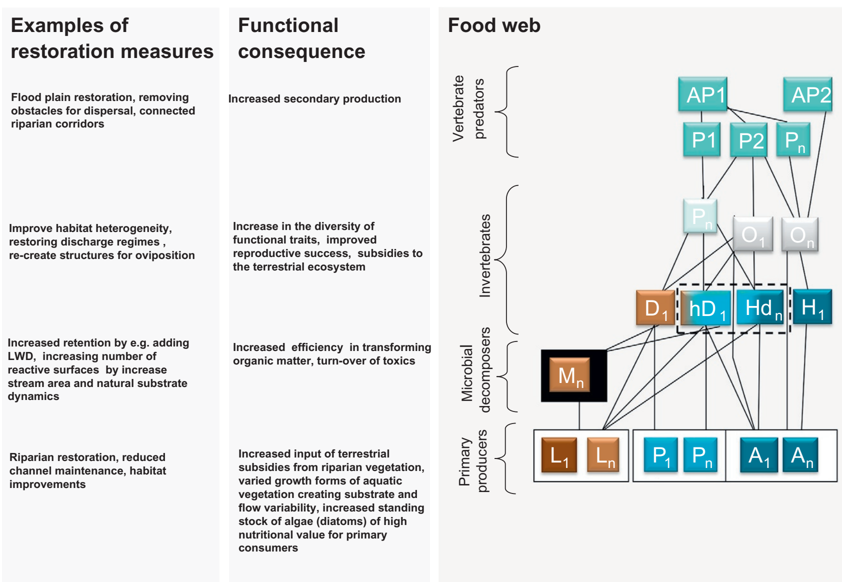


Fig. 6 Different restoration measures will influence both individual trophic levels as well as the entire food web. The figure shows examples of the possible effects on ecosystem functioning of different restoration measures (*left* and *middle panels*) with a schematic freshwater food web mapping possible effects onto trophic position. *AP*, apex vertebrate predator such as fish, amphibians and birds; *C*: carnivorous invertebrate; *O*: omnivorous invertebrate; *D*: detritivore; *hD/Hd*: herbivore–detritivore; *H*: herbivore; *AH*: aquatic hyphomycetes; *L*: leaf-litter; *P*: plant; *A*: algae.

Evaluation of restoration projects, using a space-for-time substitution approach, showed that despite plenty of suitable habitat macroinvertebrate diversity was still lower than in reference streams that had never been significantly modified (Pedersen et al., 2014). Likewise, the influence of other pressures on the system needs to be less important local physical habitat conditions (Fig. 7). Several empirical studies indicated that large-scale pressures related to catchment land use can be more important in shaping macroinvertebrate and fish communities compared to pressures at smaller spatial scales like local hydromorphological alterations (Roth et al., 1996; Stephenson and Morin, 2009; Sundermann et al., 2013), and might even limit macroinvertebrate and fish assemblages (Bryce et al., 2010; Kail et al., 2012; Wang et al., 2007). In addition, Harding et al. (1998) showed a clear land-use legacy, where catchment land use more than 3 decades back determined present day diversity of macroinvertebrates and fish.

Lastly, we have to consider that most studies that have revealed links between biota and local physical conditions have been scientific studies employing very specific sets of methodology. In reality, most assessment of ecological conditions in rivers, and effect studies of restoration interventions, are undertaken as part of monitoring programmes using sampling

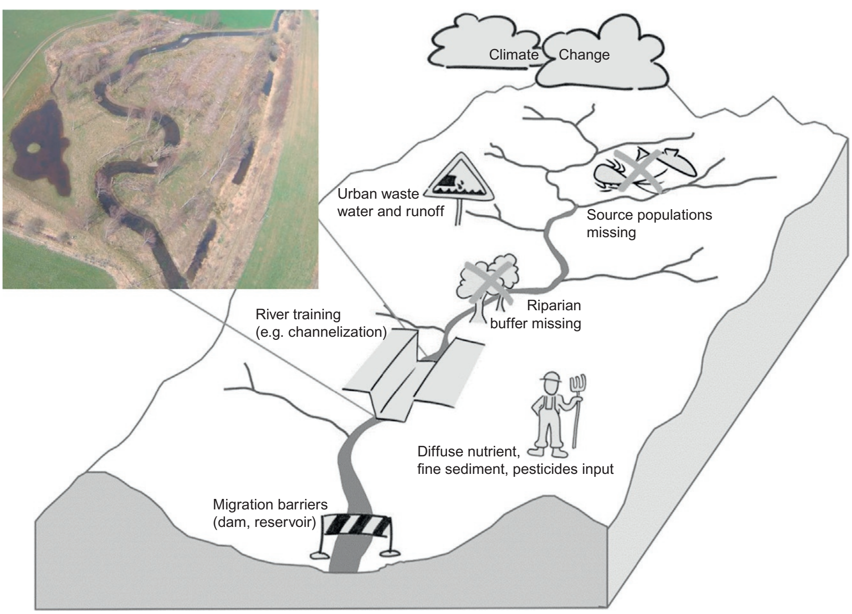


Fig. 7 Effects of local-scale restoration measures such as the remeandering of short river reaches will be dependent on other pressures acting on the catchment. Poor water quality relating to upstream land use will potentially mask any benefits of improved habitat conditions as will dispersal barriers such as dams or lack of source populations.

approaches based on CEN standards and metrics (in Europe) that have been intercalibrated. These may lack the sensitivity required to adequately monitor hydromorphology, reflecting the original development of this methodology was developed to be primarily sensitive to water quality (Friberg, 2014). These observations are important to consider when exploring links between river biota and local physical conditions in the context of restoration.

2.2 Importance of Local Physical Habitat Filters in Structuring Stream Biota

River discharge maintains riverine processes and mediates connectivity and is often the focal point of many restoration projects; however, species do not directly respond to discharge but rather flow velocities and stream power (O'Hare et al., 2011; Statzner et al., 1988). Flow velocity and stream power provide species and size-specific thresholds for habitat utilisation. In addition, the interaction between flowing water and the size and quantity of available sediment leads to diverse substrate calibres emerging from flow-induced sorting. Therefore, the most important components of local physical habitat filter in stream and rivers relate to flow conditions and substrate composition. These are interlinked as flow determines substrate. Far too often, however, substrate is introduced as a restoration measure without considering flow or stream power and this will not provide sufficient habitat conditions (Pedersen et al., 2014). In a comparative metaanalysis of 80 interacting hydromorphological processes and variables aiming to identify the most relevant factors controlling ecological degradation and restoration, Lorenz et al. (2016) identified water flow as the most important process. Accordingly, species with life history traits and ecological characteristics that are directly linked to flowing water and the related processes of sediment erosion, transport and sorting should be diagnostic in their response to hydromorphological degradation and rehabilitation. A challenge, however, is that the biotic response to high flow velocities and shear stresses is not very specific. Common thresholds values and ranges of flow velocities reported in the literature underpin this point (Fig. 8): <0.3 m/s for species rich, diverse macrophyte communities (Janauer et al., 2010), 0.3–1.0 m/s for rheophilic invertebrates (Söhngen et al., 2008; Statzner et al., 1988), and 0.1 and 0.5 m/s for hatchlings and juvenile fish, respectively (Wolter and Arlinghaus, 2003, 2004). However, these thresholds vary widely within taxa and genera, e.g. between <0.8 and >2.0 m/s for various gastropods, selected dipterids and some beetles, respectively (Statzner et al., 1988). While upper flow thresholds selects for few species adapted to a life in high

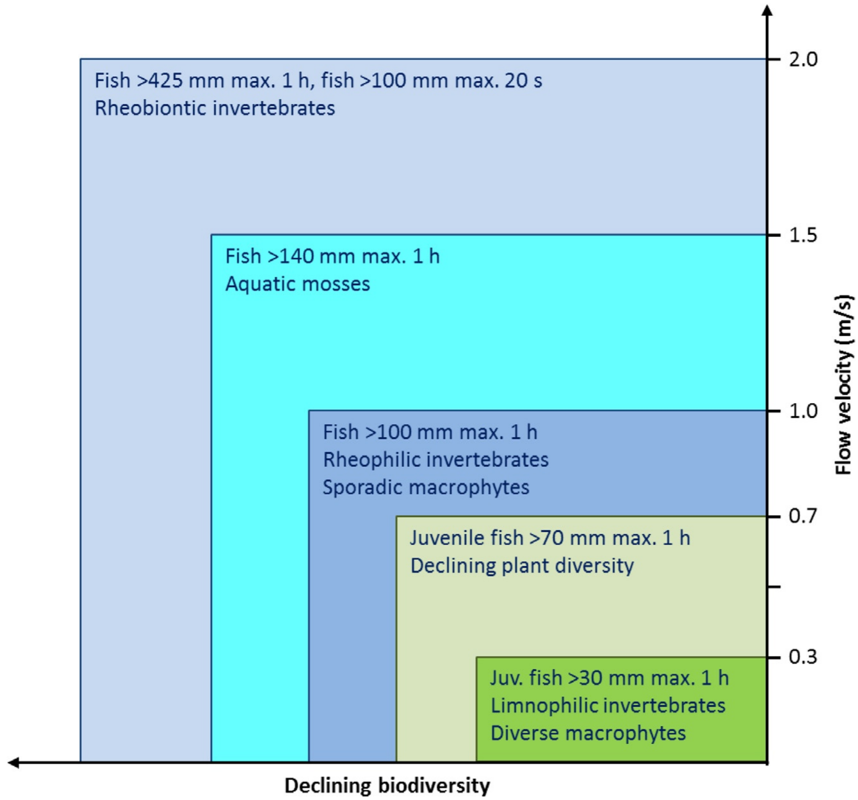


Fig. 8 Physical thresholds for diverse aquatic taxa in response to flow velocity. *Modified from Söhngen, B., Koop, J., Knight, S., Rythönen, J., Beckwith, P., Ferrari, N., Iribarren, J., Kevin, T., Wolter, C., Maynord, S., 2008. Considerations to Reduce Environmental Impacts of Vessels. Brussels: PIANC, PIANC Report 99.*

water velocities; the provision of low flow habitats <0.3 m/s similarly supports nearly all taxa, stressing the importance of these low flow habitats to sustain and improve biodiversity in restoration projects.

Benthic algae are particularly prone to the impact of increased fine sediment loads (Jones et al., 2014) and this is probably the most important type of hydromorphological stress for this organism group. A direct first principle effect is a decrease in light with increased turbidity but the most profound effect of fine sediment is the smothering of substrata to which benthic algae attach. These relatively unstable deposits (compared with larger particles) are not suitable for the attachment of long-lived sedentary species. Hence, non-motile, and particularly chain-forming taxa, cannot establish easily, further pushing the assemblage towards single-celled and motile taxa. A shift in assemblage composition towards motile taxa can be seen even where larger particles are covered with a layer of fines (Dickman et al., 2005). The lack of stability in patches, where easily erodible fine sediments accumulate, tends to result in reduced taxon richness and biomass compared to more stable patches (Biggs et al., 1999; Biggs and Smith, 2002; Matthaei et al., 2003). When comparing across streams, those with stable bed sediments support a higher biomass of diatoms than those that have unstable beds (Biggs, 1996; Biggs et al., 1999; Biggs and Smith, 2002; Jowett and Biggs, 1997). Further negative effects of hydromorphology could be expected through both direct and indirect impacts on the substrate on which benthic algae grow. Reductions in flow velocity, for example caused by impoundments, would tend to increase the deposition of fine sediment altering both bed substrate and the potential for planktonic algae to thrive. Direct modification of in-stream and marginal habitat has the potential to alter the substrate on which benthic algae grow. In restoration context, it is needed not only to provide coarse substrates for diatoms and other benthic algae to grow but to control, in particular, the delivery of fine sediments to the river system.

Water flow and turbulence determines the macrophytes of running waters, governs plant form, dominates the growth-controlling factors and defines the habitats (Biggs, 1996; Dawson, 1988; Folkard, 2011; Schutten et al., 2005). Generally, plants with high drag coefficients and low anchoring strengths are those species most susceptible to high velocities (Biggs, 1996). Therefore, the ability of a plant to tolerate water movement without suffering mechanical damage relies either on minimising the hydrodynamic forces or maximising its breakage and uprooting strengths (Bornette and Puijalon, 2011). Emerged growth is considered favourable under low flow conditions, especially since this growth form is not affected by any reduction of light due to the attenuation by the water column (Bal et al., 2011). Submerged growth is supported by rougher conditions and high flow velocities due to the flattening, compressing and reconfiguration of plants which lower the drag forces (O'Hare et al., 2007; Puijalon et al., 2005; Sand-Jensen and Pedersen, 2008; Sukhodolova, 2008). Slightly enhanced flow velocities promote the growth of aquatic macrophytes due to facilitated diffusion of CO_2 and nutrients (Madsen et al., 2001) with increases in vegetation evident at velocities up to 0.3 m/s with a peak at about 0.3–0.5 m/s depending on the species (Janauer et al., 2010; Riis and Biggs, 2003). At higher velocities, plant biomass and diversity decrease and flow velocities higher than 0.8 m/s dislodge and eliminate most in-stream macrophytes (Bernez et al., 2007; Chambers et al., 1991; Janauer et al., 2010; Madsen et al., 2001). Macrophytes have a large potential as ecoengineers in river restoration due to their ability to changing flow and substrate conditions at the reach scale.

However, macrophytes are important primarily in low energy systems in the lowlands.

Substrate composition influences benthic macroinvertebrate communities, in particular the quality and quantity of organic matter in sediment and the stability of the substrate (Buss et al., 2004; Jowett, 2003; Maxted et al., 2003; Timm et al., 2008). Density and diversity of macroinvertebrates show a general increase with substrate particle size (Beauger et al., 2006; Duan et al., 2009; Reice, 1980) with the exception that high organic content can sustain large numbers of certain macroinvertebrate taxa on fine-grained substrates (Pan et al., 2012). The hydraulic environment and in particular bed shear stress is a good predictor of benthic macroinvertebrates distribution at the microhabitat scale, because it accounts for the turbulences at the bed surface generated by sediment roughness and its associated drag and lift forces (Mérigoux and Dolédec, 2004; Möbes-Hansen and Waringer, 1998). Consequently, Dolédec et al. (2007) and Mérigoux et al. (2009) were able to characterise shear stress preferences of 181 benthic macroinvertebrate taxa using the methodology of Statzner et al. (1991). Only few studies provide velocity dislodgement thresholds for benthic macroinvertebrates (e.g. Holomuzki and Biggs, 2000; Statzner et al., 1988; Wilzbach et al., 1988). The average shear stress necessary to detach and dislodge macroinvertebrates has been estimated for a total of 27 taxa (Borchardt, 1993; Gabel et al., 2012; Hauer et al., 2012). However, responses of macroinvertebrates to hydraulic conditions and shear stress at the microhabitat scale are not independent of larger scale features such as stream size (Dolédec et al., 2007; Hauer et al., 2012; Mérigoux et al., 2009). In addition, critical shear stress thresholds for the same taxa vary between studies (Gibbins et al., 2010; Hauer et al., 2012) and more recently Gibbins et al. (2016) suggested water velocity and Froude numbers are better predictors of macroinvertebrate drift than shear stress.

To identify species whose presence is not primarily determined by substrate preference or hydraulic preference, we plotted preferred substrate fractions reported by Tolkamp (1982) and Singh et al. (2010) in classes from 0 (128–256 mm) to 11 (0.125–0.05 mm) against shear stress calculated from Dolédec et al. (2007) and Mérigoux et al. (2009) (Fig. 9). Outliers above and below the substrate–shear stress relationship obtained are characterised by significantly higher shear force tolerance compared to the preferred substrate calibre and vice versa. When restoring rivers, it is therefore important to acknowledge that macroinvertebrates do respond to shear stress and not just

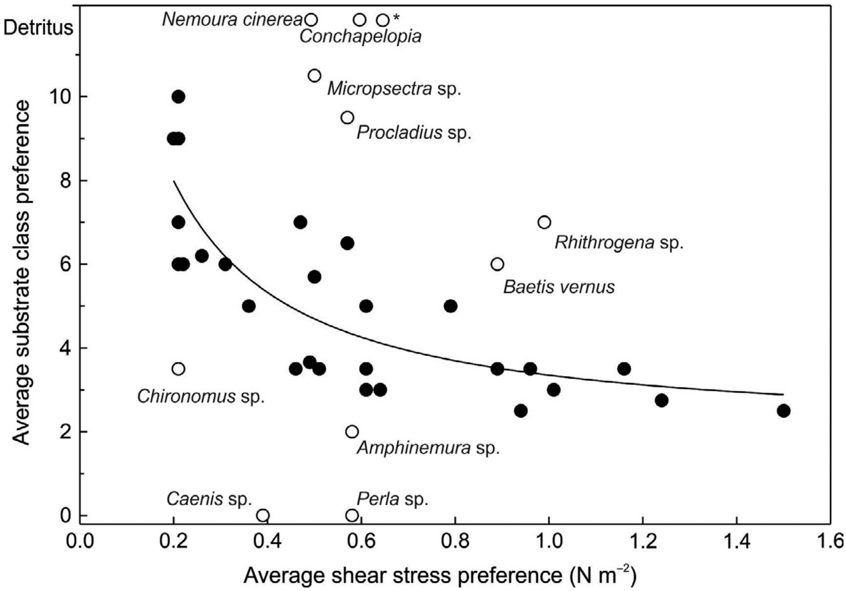


Fig. 9 Regression model of benthic macroinvertebrates substrate preferences in categorical size classes from class 0 (128–256 mm) to 11 (0.125–0.05 mm) plotted against their hydraulic preferences. Outliers (*white circles*) indicate taxa not primarily determined by hydromorphology (* = *genera Brillia*, *Corynoneura*, *Diplocladius*, *Eukiefferiella*, *Rheocricotopus*).

substrate sizes. The focus must be on restoring natural fluvial processes that creates a range of different shear stress microhabitats both in space and in time. It also evident that range in response to both shear stress and substrate size of most taxa is fairly wide, calling for the creation of a highly variable physical environment.

Fish are comparably long living, mobile organisms with various habitat requirements, habitat shifts during ontogeny, and functional differences between age groups and hence utilising habitats at various spatiotemporal scales (Wolter et al., 2016). Coarse, well-oxygenated and permeable gravel beds are essential for fish species that are depend on such substrates for spawning and gravel spawning are commonly considered as adaptation of fish to faster flowing environmental conditions by protecting eggs and hatchling from becoming washed away (Balon, 1975; DeVries, 1997; Jungwirth et al., 2000). Significant empirical relations between fish length and gravel diameter have been reported for red digging salmonids (Crisp, 1996; Kondolf and Wolman, 1993). Headwater species, such as trout and grayling, lay their eggs into the substrates at depths from a few centimetres and up to 30 cm and, in addition to sediment size, they are dependent on permeable sediment with interstitial flow of oxygen-rich water (DeVries, 1997; Riedl and Peter, 2013; Sternecker et al., 2013). Accumulation of fines <1 mm has been reported causing most significant impacts on hatch and survival of fish larvae even at rather low proportions (Heywood and Walling, 2007; Julien and Bergeron, 2006; Soulsby et al., 2001). As with other aquatic organisms fish are exposed to flow velocity and stream power, which both set physical thresholds for habitat utilisation. Swimming performance of fish has been reviewed and analysed as a proxy for their ability to withstand absolute physical forces set by flow velocity (Wolter and Arlinghaus, 2003, 2004). The individual swimming performance depends on species, swimming mode, size, temperature, ontogenetic stage, photoperiod, oxygen tension, pH, salinity and various pollutants and toxins, with total length as paramount trait (reviewed in Hammer, 1995; Videler, 1993; Wolter and Arlinghaus, 2003). Based on the metaanalysis of 168 swimming performance studies covering 75 freshwater fish species, Wolter and Arlinghaus (2003) derived general models for burst and critical swimming performance of fish. The general models of length-specific burst and critical swimming performance were highly significant. As expected, salmonids exhibited the highest burst swimming performance; however, the differences detected between the small-sized individuals of different taxonomic orders were not significant (F-test, $p=0.142$). Thus, the swimming performance model applies for all fish up to 60 mm total length, which is important as one would intuitively think that rheophilic fish perform superior to eurytopic and limnophilic fish. Consequently, a 56 mm long fish already maintains a speed of 1.0 m/s for 20 s, but only 0.54 m/s in the critical mode for 1 h (Fig. 10).

In conclusion, the different organism groups found in rivers will respond to different components of the physical environment and some links to restoration are straightforward in substrate preferences of gravel spawning fish as well as critical swimming speeds. In both cases, this knowledge have been used in restoration measures such as instalment of riffles and by adjusting current velocity in bypass streams (at dams and weirs) to allow migration and resting of specific fish species. However, we also show that there is a generally high variability in response and that physical habitat preferences are quite nonspecific in a way that limit our ability to be very prescriptive in the approach to restoration, also considering the complex interaction between hydromorphological degradation and environmental factors.

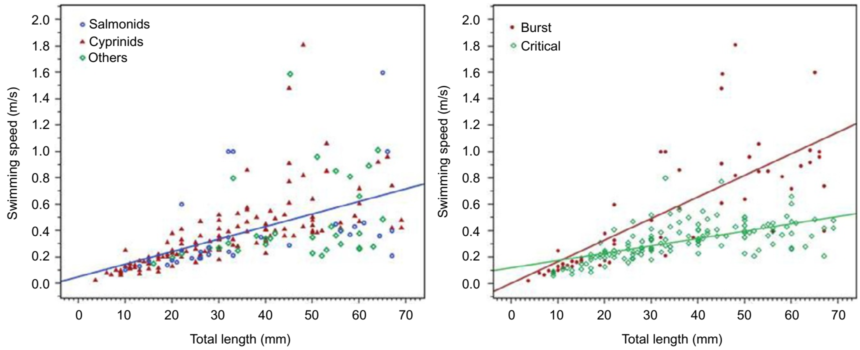


Fig. 10 Burst swimming performance (*left*) of salmonids, cyprinids and other fish species up to 60 mm total length compared to their critical swimming performance (*right*). Regressions did not significantly differ between families (*F*-test, $p = 0.142$) and followed the models $U_{burst} = 0.0068\ TL^{1.24}$ (df = 84, $R^2 = 0.83$; $p < 0.001$) and $U_{crit} = 0.0067\ TL^{1.09}$ (df = 155, $R^2 = 0.60$, $p < 0.001$). *Modified from Wolter, C., Arlinghaus, R., 2003. Navigation impacts on freshwater fish assemblages: the ecological relevance of swimming performance. Rev. Fish Biol. Fish. 13, 63–89.*

2.3 Are We Capable of Detecting Impacts of Hydromorphology on Biodiversity Using Standard Methods?

Several recent studies have revealed weak relationships between a standardised measure of habitat quality (River Habitat Survey (RHS)) and a number of macroinvertebrate indices in streams along a hydromorphological degradation gradient (e.g. Vaughan et al., 2009). Existing monitoring datasets collected across Europe, to assess ecological status as stipulated by the WFD, were analysed as part of the REFORM project and we found in general weak statistical relationships between identity-based (taxonomic) ecological indicators and measures of hydromorphological quality as shown in Table 2. In most cases, other impacts such as eutrophication caused by agriculture had a much more pronounced impact on the biological indicators compared with changes to the physical environment, underpinning the importance of considering all pressures together when restoring rivers.

Metrics developed to detect hydrological impairment and hydromorphological degradation were not more discriminative to other types of metrics when tested on a Danish dataset that included a hydrological time series (Table 3). Some of the strongest relationships identified were between macroinvertebrate metrics that measure the presence of taxa with a preference for specific flow or substrate type conditions and between indicators of

Table 2 Spearman Correlation Coefficients of EQRs (Standardised Ecological Quality Ratios) of Diatoms, Benthic Macroinvertebrates and Fish, and the Mean EQR of the Three Groups, Against the PCA-Gradients from Finnish Streams with the Main Environmental Gradients

	EQR Diatoms	EQR Macroinvertebrates	EQR Fish	Mean EQR
Number of sites	132	139	96	91
PCA1 (agriculture)	**−0.550**	**−0.563**	**−0.488**	**−0.670**
PCA2 (morphological alteration)	−0.060	0.023	−0.067	−0.025
PCA3 (urban)	−0.064	−0.086	0.154	0.151
PCA4 (naturalness)	**0.186**	0.097	0.065	0.121

Biological and environmental monitoring data are from 150 river sites across Finland (not all biological groups were present at each site for direct comparison). Significant coefficients are given in bold (for details on results, please consult http://reformrivers.eu/results/effects-of-hydromorphological-changes).

Table 3 The Relationship of Selected Macroinvertebrate Metrics Sensitive to Different Types of Stress and Hydrological Data Shown as Pearson Correlation Matrix for Maroinvertebrates Metrics[a] and Flow Statistics[a]

Sensitive to	MESH HYMO Stress	ASPT Organic Pollution	LIFE Low Flow	EPT General Degradation	SPEAR (%) Pesticide Exposure
Q90	0.61	0.59	0.52	0.44	0.6
Q10	−0.58	−0.52	−0.47	−0.43	−0.55

[a]*MESH*, Maroinvertebrates of Estonia: score of hydromorphology; *ASPT*, average score per taxon; *LIFE*, Lotic-Invertebrate Index for flow evaluation; *EPT*, the total number of taxa belonging to Ephemeroptera, Plecoptera, Trichoptera order; *SPEAR* (%), indicated toxicity of pesticides and organic pollution in water; Q10, flow magnitude exceeded for 10% of the time; Q90, flow magnitude exceeded 90% of the time.
All selected macroinvertebrate metrics revealed a significant relationship ($p<0.001$) with flow statistics (Q90 and Q10) and they all exhibited a similar direction of response at Q90 (positive) and Q10 (negative).

these specific conditions, which reflects the potential of using trait-based metrics to evaluate hydromorphological conditions. However, a subsequent analysis revealed few traits in macroinvertebrates that potentially could distinguish between hydromorphological and other stressors although relationships were not very significant.

The finding using larger monitoring datasets is somewhat in contrast with the literature reviewed at the beginning of this chapter that showed that both substrate composition and shear forces influenced macroinvertebrate community composition. The reasons for the lack of sensitivity might be most likely attributed to: (1) a number of explanatory variables not being measured as part of routine biological monitoring programmes or (2) the hydromorphological assessment schemes that do not necessarily register elements of importance to the in-stream biota. As an example, riffle habitats that are visually assessed as identical can differ markedly in macroinvertebrate diversity and community composition (Pedersen and Friberg, 2007). This very clearly suggests that human perception is biased, leading to subjective assessments that mask relationships between biota and hydromorphology and drives restoration efforts in a direction which does not optimise ecological recovery.

No effects of hydromorphological degradation on indices based on phytobenthos were detected from analysis on a large-scale dataset with no significant effects detected of channel impairments such as straightening (Fig. 11).

However, indices developed to assess eutrophication stress (e.g. Trophic Diatom Index (TDI) and related indices) appear robust to hydromorphological alteration as there was a significant relationship with $\log_{10}$ orthophosphate for almost all indices. With regard to fine sediment stress specifically, analysis of a spatial dataset across England and Wales showed a strong relationship between fine sediment and the diatom community composition, which suggests that diatoms could be used as an indicator of fine sediment stress. In contrast to this finding, and more in support of general analysis of the response to hydromorphological stress, phytobenthos community composition was primarily impacted by soluble reactive phosphorous concentration in experiments. Fine sediment treatment did not significantly impact on either chlorophyll-a concentrations or ash-free dry weight, suggesting that algal growth was unaffected.

Macrophyte trait characteristics changed significantly in response to hydromorphological degradation in small streams. Several traits could be identified such as species growing from single basal meristems declined and that species with a high overwintering capacity increased. However, with one exception, the trait heterophylly (the ability to have different types of leaves above and below water), impacts of hydromorphological stress could not be separated from eutrophication. More traits were specific to eutrophication, which could aid in the diagnostic of main stressors in multiple stress systems. Although macrophytes are very important in some river types, they have limited applicability as a general indicator for the range of European river types. For lowland streams, which are the type most often

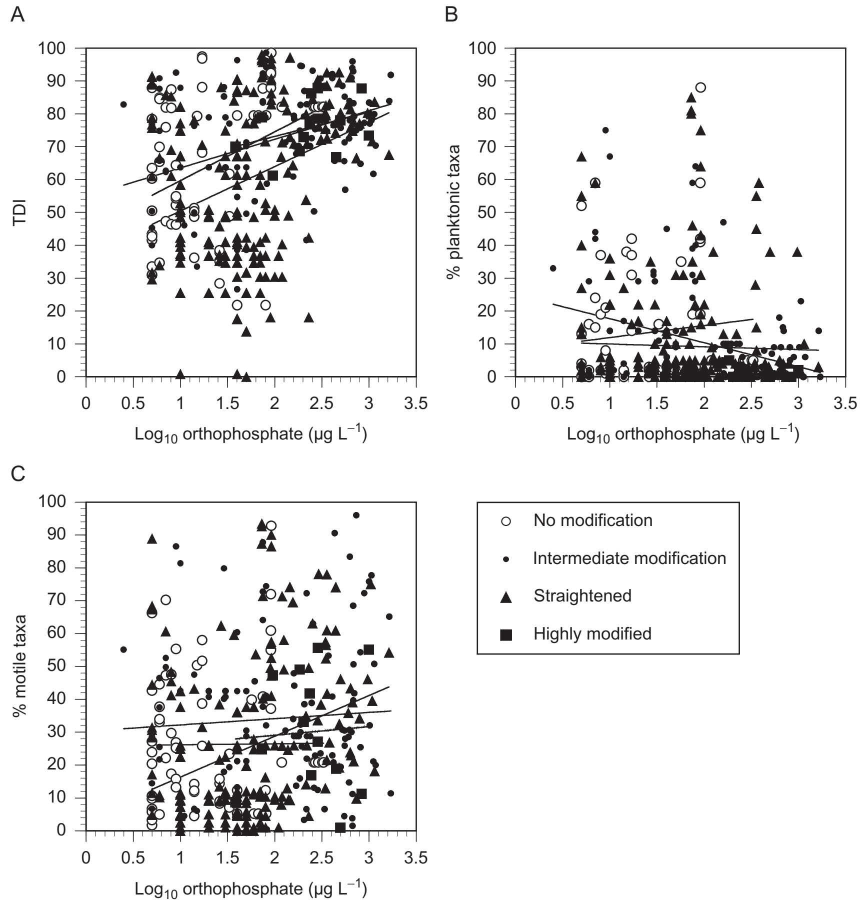


Fig. 11 Influence of channel modification on the relationship between $\log_{10}$ orthophosphate concentration and (a) Trophic Diatom Index (TDI)—an indicator of eutrophication (phosphate) and two ecological species traits: (b) % planktonic taxa and (c) % motile taxa.

physically modified, they could be used if a trait-based metric was developed (Baattrup-Pedersen et al., 2015).

Forty-nine per cent of the studied European freshwater fish species in REFORM showed a significant response to hydromorphological stress. Using conceptual models that link pressures via processes to responses, it was identified that benthic fish like Cobitidae, Balitoridae, Cottidae or Gobiidae showed the most consistent response to HYMO stress. This response can be related to their dependence on substrate dynamics and low mobility. Conceptual models should be viewed as a first step, and although more traditional metric approaches using fish also has significant potential, both approaches require significant development before they can be applied in regular monitoring. The inherent difficulty in fish methods based on number of taxa is that their use is restricted to stream/river types with a minimum of species diversity, leaving out most small streams. In these types more detailed investigations on length frequency distributions can be related to HYMO stress. Many HYMO pressures (but also non-HYMO pressures) generate a clear response in the fish community size structures, particularly inducing changes in the overall shape of the size spectra, which might therefore be useful as a new potential metric for assessing impacts of HYMO stress.

Key to the strength of these relationships was sampling methods and to some degree organism size, with fish showing most promise which is not surprising as they, due to their generally large body size, integrate over larger spatial scales and thereby reduce the variability introduced when assessing small-scale features such as individual microhabitats. Relationships were further improved when using species traits (also using macrophytes) that are more closely linked to the habitat template, including hydromorphology. These findings suggest that scales of sampling are a core issue in assessing the effects of restoration measures and methods using species traits of large(r) organisms are likely to be the most sensitive. However, the focus on in-stream biota that is routinely monitored ignores that many of the pronounced effects of degraded hydromorphology relates to the riparian zones and the wider floodplain. When comparing restored and nonrestored river sections, riparian ground beetles most strongly responded to hydromorphological improvements, followed by fish and floodplain vegetation (Hering et al., 2015).

2.4 Which Are the Best Standard Indicator to Detect HYMO Stress and Recovery Through Restoration?

Our findings leave water managers with a significant challenge when diagnosing the reason for not obtaining GES in a waterbody or detecting positive effects of a restoration intervention, aiming at improving the physical environment. Even for fish, the organism group which showed most promise, there is a significant amount of work to do before sensitive metrics to hydromorphological change can be applied in water management. The issue with the impact of multiple stressors on the biota has been shown to preclude our ability to disentangle hydromorphological change from other stressors. It appears from the analysis undertaken that eutrophication is a stronger driver of community changes. This is, however, most likely also related to the quality of the hydromorphological assessments, which in most of the larger datasets were fairly superficial in the current analysis.

3. THE CURRENT RESTORATION PARADIGM

3.1 The Many Ways of Restoring Rivers

In recent decades, Europe has made significant progress in reducing pollution and water quality has significantly improved, albeit there are still issues relating to diffuse pollution from agriculture and point-source discharges from wastewater treatment plants (EEA, 2012). Concurrently, with the implementation of the WFD, it has been mandatory to set environmental targets focused on ecological status of surface waters as well as good chemical water quality. Hence, despite the significant improvements to water quality and some biological indicators, the continued trend of declining freshwater biodiversity is unacceptable from a legislative point of view (Aarts et al., 2004; SCBD, 2010). Accordingly, a shift in restoration paradigm has occurred from a focus on purely physicochemical water quality to a paradigm encompassing ecological quality, hydromorphological and habitat conditions. Both hydromorphological pressures and altered habitats have been reported as the most common impact for 48.2% and 42.7% of all river water bodies, respectively (Fehér et al., 2012). Hydromophology was considered a larger pressure on water bodies than diffuse pollution across the full range of arable land use, in particular in intensively farmed catchments reflecting the impact of drainage (Fig. 12).

Despite numerous hydromorphological restoration projects implemented since the 1940s, the number of projects has exponentially risen in the last 10–15 years (e.g. Alexander and Allan, 2006; Bernhardt et al., 2005; Feld et al., 2011; Roni et al., 2008). Here we have compiled a database of 813 hydromorphological river restoration projects described in 878 publications, 53% of them implemented after 2003 and published after 2007. Most of the projects (649) were from Europe. The review revealed a range in the number of river hydromorphological elements that have been addressed as well as the employment of a substantial variety of 53 specific measures in total (Table 4). Especially, the improvement of habitat structures in the stream channel itself has been targeted with a number of different measures, most common the removal of artificial embankments, the addition of large wood and the provision of spawning gravel. In-stream measures typically address specific hydromorphologic elements, not the underlying processes. In contrast,

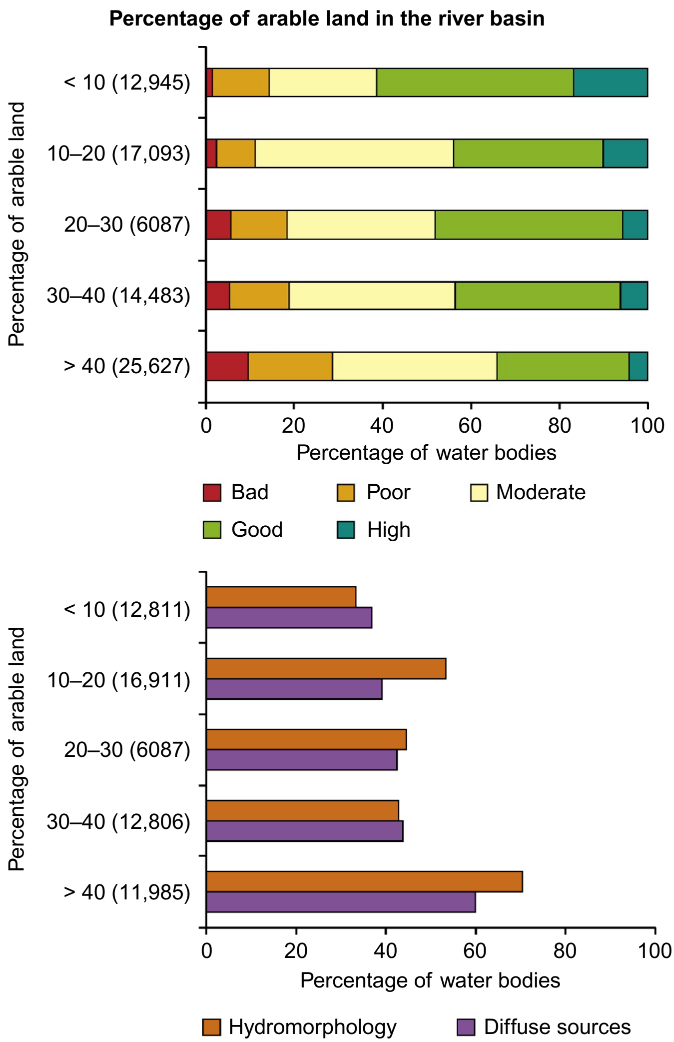


Fig. 12 Assessment of ecological status in relation to percentage arable land in the catchment with the main pressures on aquatic ecosystems listed. Hydromorphology is perceived as a larger pressure than diffuse pollution in most cases. *Figure obtained from the EEA (first published 2012) at: http://www.eea.europa.eu/data-and-maps/figures/ecological-status-potential-and-pollution/ecological-status-potential-and-pollution.*

improvement of the riparian zone is primarily addressed by vegetated buffer strips. Remeandering and hydrological reconnection of floodplain water bodies are the most commonly applied intervention to improve river planform and floodplains, respectively (Table 4). The largest restoration measures, most often river widening, have been implemented in the lower parts of mountain

Table 4 Number of Hydromorphological Restoration Measures Reportedly Implemented in the 813 River Restoration Projects Surveyed

Restoration Measure	**Number**
Water quantity/Hydrology	*85*
Reduce water surface water abstraction without return	4
Improve water retention (e.g. on floodplain, urban areas)	59
Reduce groundwater abstraction	6
Improve/create water storage (e.g. polders)	7
Increase minimum flow (to generally increase discharge in a reach or to improve flow dynamics)	16
Water diversion and transfer to improve water quantity	3
Recycle used water (off-site measure to reduce water consumption)	4
Reduce water consumption (other measures than recycling used water)	1
Sediment quantity	*47*
Add/feed sediment (e.g. downstream from dam)	21
Reduce undesired sediment input (e.g. from agricultural areas or from bank erosion other than riparian buffer strips!)	8
Prevent sediment accumulation in reservoirs	1
Improve continuity of sediment transport (e.g. manage dams for sediment flow)	9
Trap sediments (e.g. building sediment traps to reduce washload)	16
Reduce impact of dredging	3
Flow dynamics	*60*
Establish environmental flows/naturalise flow regimes (does focus on discharge variability)	36
Modify hydropeaking	7
Increase flood frequency and duration in riparian zones or floodplains	11
Reduce anthropogenic flow peaks	6
Shorten the length of impounded reaches	3
Favour morphogenic flows (could also be considered a measure to improve planform or in-channel habitat conditions)	3
Longitudinal connectivity	*143*

Table 4 Number of Hydromorphological Restoration Measures Reportedly Implemented in the 813 River Restoration Projects Surveyed—cont'd

Restoration Measure	**Number**
Install fish pass, bypass, side channel for upstream migration	32
Install facilities for downstream migration (including fish friendly turbines)	25
Manage sluice, weir and turbine operation for fish migration	18
Remove barrier (e.g. dam or weir)	103
Modify or remove culverts, syphons, piped streams	13
In-channel habitat conditions	*560*
Remove bed fixation	77
Remove bank fixation	144
Remove sediment (e.g. mud from groyne fields)	30
Add sediment (e.g. gravel)	102
Manage aquatic vegetation (e.g. mowing)	40
Remove or modify in-channel hydraulic structures (e.g. groynes, bridges)	33
Creating shallows near the bank	106
Recruitment or placement of large wood	130
Boulder placement	115
Initiate natural channel dynamics to promote natural regeneration	112
Create artificial gravel bar or riffle	128
Riparian zone	*218*
Develop buffer strips to reduce nutrient input	18
Develop buffer strips to reduce fine sediment input	13
Develop natural vegetation on buffer strips (other reasons than nutrient or sediment input, e.g. shading, organic matter input)	186
River planform	*321*
Remeander water course (actively changing planform)	156
Widening or rebraiding of water course (actively changing planform)	115
Shallow water course (actively increasing level of channel bed)	54
Narrow overwidened water course (actively changing width)	25

Continued

Table 4 Number of Hydromorphological Restoration Measures Reportedly Implemented in the 813 River Restoration Projects Surveyed—cont'd

Restoration Measure	Number
Create low-flow channels in oversized channels	38
Allow/initiate lateral channel migration (e.g. by removing bank fixation and adding large wood)	31
Create secondary floodplain on present low level of channel bed	8
Floodplain	*320*
Reconnect existing backwaters, oxbow lakes, wetlands	158
Create seminatural/artificial backwaters, oxbow lakes, wetlands	107
Lowering embankments, levees or dikes to enlarge inundation and flooding	64
Back-removal of embankments, levees or dikes to enlarge the active floodplain area	23
Remove embankments, levees or dikes or other engineering structures that impede lateral connectivity	32
Remove vegetation	85
Others	*243*

Several measures and measure combinations per project possible; note, connectivity measures were only counted in combination with other hydromorphologic measures. The database excludes fish migration facilities. Fish passes as connectivity measures are prerequisite for fish dispersal and may provide access to restored habitats, but they will rarely improve overall hydromorphological and habitat conditions.

rivers, where typically only one pressure was reported relating to channel narrowing and embankments (Kail and Wolter, 2011).

Generally, evaluations of the outcome of projects are still rare (Bernhardt et al., 2005; Kail et al., 2015; Palmer et al., 2010) despite the increasing number of restoration interventions and an increased societal drive to identify efficient solutions that have economic benefits (Everard, 2012; Reichert et al., 2015; Smith et al., 2014).

3.2 Restoration Measures Are Dependent on River Type

In more detail, we found that restoration effort was dependent on river type across Europe with marked differences between lowland rivers (<200 masl.), lower mountain rivers (<800 masl.), upland and glacial rivers and large rivers (catchment > 10,000 km^2).

Floodplain wide measures in lowland areas, which are predominantly used for agricultural production across Europe, will most often necessitate either buying land off farmers or compensating them for a change into a less profitable land use as increased inundation and ground water tables limits the commercial value of the land. In reality costs of land at the floodplain scale are so high in lowland areas that this type of restoration is rarely executed. Naturally lowland river systems are meandering or have multiple channels configuration (anastomosing) with the active channel covering large parts of the valley (Rinaldi et al., 2016). The most frequently applied measure is lowering the floodplain in combination with a shallow stream bed whereby the stream can shape the floodplain, rewet it and form a single channel with sometimes a secondary channel. Restoring an anastomosing planform, or allowing natural features of lowland systems such swamp forest to develop, often meets resistance from other users of the floodplain. Most of the measures in lowland rivers aim to restore the channel planform (56%), primarily by remeandering previously channellised stretches of river or some intermediate form of channel plan modifications, e.g. digging a two-stage profile. More often these channel planform measures are combined with in-channel measures, like removal of bank fixation and/or adding local structures such as groynes. Probably this is because of the low cost of in-channel measures compared to changes in channel planform that needs adjacent land. In theory, remeandering will affect in-channel habitat conditions but sand-bed rivers have a limited potential for recruiting coarse substrates. They are to a large degree reliant on woody debris and plants to create heterogeneous habitats. Our review of projects carried out in lowland areas furthermore showed that active remeandering of lowland rivers can also decrease microhabitat diversity, i.e. there were cases where remeandering led to a decrease in river velocity resulting in particulate organic material as the main microhabitat, while in the unrestored section more habitats were present. Infrequently, projects will also involve reconnection of former secondary channels or the creation of new where original are lost. Sometimes these 'reconnection' measures are combined with in-channel measures including removal of bank fixation and/or adding local structures such as tree logs. Restoration of the riparian zone in lowland areas is always limited to local areas where rewetting and flooding is a possibility. Often water safety arguments are behind creating areas that can get inundated and store water to reduce risk of downstream flooding. The most frequently applied measure in lowland rivers is reconnecting old meanders and oxbow lakes; removing weirs and restoring river banks. Restoration of the riparian zone is always combined with channel planform and in-channel measures and the extent in term of width restored is often limited.

Most of the restoration projects in single thread, lower mountain rivers applied in-channel measures to increase habitat complexity (75–80%), most frequently by removing bed and bank fixation, adding large wood and boulders and creating shallow slow-flowing areas, while measures to explicitly restore natural sediment dynamics (e.g. by adding sediment, restoring natural sediment transport or limiting fine sediment input) were rarely applied (1–6%). Many projects also aimed to restore a more natural planform (40–54%), e.g. by widening or remeandering the stream channel, or involved developing a riparian buffer strip (~30%) or restored floodplain habitats (~48%).

The majority of measures taken in mountain rivers and glacial rivers with single-thread channels aim to restore the flow alteration as these rivers across Europe have been dammed for energy production, drinking water supply and irrigation of farmland. Most important is the restoration of the natural flow regime, the reestablishment of the natural flow dynamics and the increase of water flow quantity in case of residual water flow not used for hydropower production. Hydropeaking (release of pulses of water), impoundment and water abstraction are relevant topics because of the dynamism of the hydropower sector and the need to mitigate and remediate adverse ecological impacts. Hydrological measures focused on mitigating the flow alteration are often applied at a local/small scale without solving the hydrological dynamics that result from catchment-wide activities. Individual measures at each hydropower plant are usually set without considering the downstream or upstream situation. In addition to restoring flow regimes, natural sediment regime and wood delivery will in some projects be restored as will in-stream habitats to mitigate the negative effects of hydropeaking. Even though the sediment regime in highland river types is usually not compromised, the building of check dams and the subsequent retention of sediment and wood can cause negative effects. These effects (e.g. increased bed and bank erosion, bed incision and negative sediment budget in wide floodplains) are visible far downstream at the lowland rivers. The input of sediment at downstream reaches is a commonly applied even though it is considered an unsustainable countermeasure. Restoring natural processes (e.g. restoration of water and sediment regime by removing blocking debris in the upper catchment) have a better effect on recovery, compared to local-scale interventions (e.g. wood or gravel addition at a lower part of the river catchment), but are rarely undertaken.

Large rivers typically serve inland navigation and have been straightened, regulated and embanked to improve their function as waterways. This designated use sets significant boundaries for river restoration, because it prevents the implementation of all kinds of measures that impact on fairway dimensions, as many of the in-channel measures do. Accordingly, the number of measures implemented is rather low compared to their dimension and the multitude of existing pressures and impacts. Most hydromorphological restoration measures implemented address the floodplain and riparian vegetation; however, by far majority of measures are conceptual. This category sums up investigations, research, pilot studies, but also changes in legislation, stakeholder involvement, information and education.

3.3 Using Current Management Plans as Indicators of Restoration Practises

In parallel to the compilation of the restoration project database, the first RBMPs and PoMs of the EU Member States were accessed for this study through the Water Information System for Europe (http://water.europa.eu). A previous assessment of the German PoMs revealed a reasonable coherent selection of measures in accordance with the analysis of pressures and impacts (Kail and Wolter, 2011). At the same time, this analysis showed a general lack of knowledge on the effectiveness of restoration measures especially those used to enhance the ecological status of lowland rivers and heavily modified water bodies.

We have translated the supplementary measures to enhance ecological status and classified them according to restoration measure groups. For these planned measures, and the preexisting restoration projects, we compared the consistency of measures suggested against identified pressures. Furthermore, we examined for any bias in the selection of measures based on available knowledge and potential ecological effects of the planned measures. In total 49,055 supplementary measures were listed for European river basin districts (17,341 for Continental Europe, i.e. excluding UK which had a more detailed list of measures than other member states). The following refers only to the supplementary measures listed for Continental Europe: conceptual measures (e.g. investigations, stakeholder information and legislation) were the most frequently planned measures, accounting for 56% of all measures followed by measures addressing water quality (18%) and hydromorphology (14%). The share of hydromorphological measures planned was dominated by floodplain rehabilitation (15%), in-stream habitat enhancement (11%), hydrology (16%), connectivity (17%) and riparian buffers (12%)

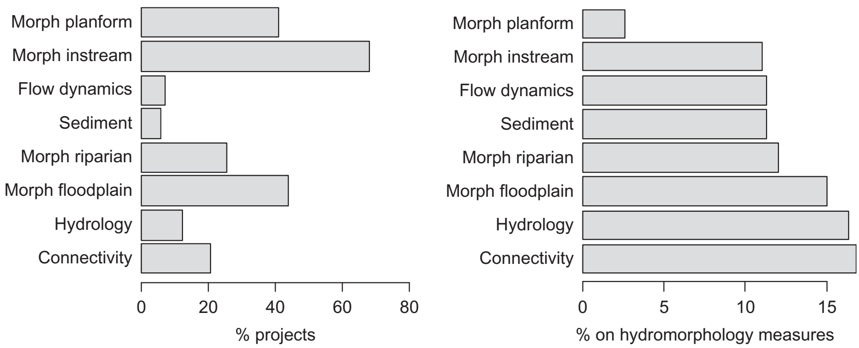


Fig. 13 Proportion of measures targeting hydromorphological improvements in 649 implemented European restoration projects (*left*) and of 2428 supplementary hydromorphological measures planned for Continental Europe river basin districts in the PoMs (*right*).

(Fig. 13, *right panel*). In comparison, in-stream habitat enhancement was the dominating measure (68%) already implemented in European rivers in the past (Fig. 13, *left panel*). Similarly, the proportion of projects addressing river planform already implemented (40%) by far exceeds those planned (<3%).

These findings provide interesting new insights: only 14% of all planned supplementary measures address hydromorphology and this is in spite of the fact that hydromorphological modifications are among the most significant pressures identified throughout Europe (EEA, 2012; Fehér et al., 2012). Furthermore, a surprisingly low percentage (11%) of the planned hydromorphological measures address in-stream habitat enhancement, although this measure retrospectively was very frequently employed (in 68% of European restoration projects in our database), implying that ample experience should be available. This obvious discrepancy between implemented restoration projects and further planning of measures raises questions as to the reasons why. One reason could be that the predominant use of in-stream measures during the first planning period have been assessed as ineffective measures, because they often provide habitat forms while ignoring the habitat forming hydromorphological processes (e.g. Roni et al., 2008; Wolter, 2010). Natural hydromorphological conditions are often only partially reestablished and the natural flooding may not be enabled which prevents the natural links between the stream and the riparian zone. Furthermore, impaired water quality due to land use (agriculture and forestry) in the catchment often prevents achieving the ecological goals of the habitat restorations.

3.4 Limitations of Current and Planned Restoration Approaches

In summary, our review of restoration projects conducted across Europe, as well as planned measures in management plans, revealed a number of limitations that might influence the effectiveness of the interventions:

- Morphological measures are very common in all river types, typically implemented as engineered solutions creating relatively static habitats at small scales with little or no interplay with geomorphic processes.
- Hydrological measures are mostly applied only locally without solving issues related to larger scale catchment-wide activities.
- Most restoration projects do not tackle the underlying processes of natural flow and sediment dynamics that acts out on larger scales.
- Even moderate water pollution and fine sediment input, as well as missing source populations for colonisation and dispersal barriers, may limit restoration effects and these catchment-scale constrains to restoration success are generally not considered.

4. EFFECTS OF RESTORATION

An increasing number of primary research studies have reported results on effects of river restoration which have been summarised in several narrative reviews (Palmer et al., 2010; Roni et al., 2002, 2008), semiquantitative reviews using vote counting (Palmer et al., 2014) and quantitative metaanalyses (Kail et al., 2015; Miller et al., 2010; Stewart et al., 2009; Whiteway et al., 2010). Moreover, there is a growing number of studies on multiple restoration projects which also allow one to draw more general conclusions, either on single organism groups (Jähnig et al., 2010; Lepori et al., 2005; Lorenz et al., 2012; Pretty et al., 2003; Schmutz et al., 2014) or comparing effects on different organism groups (Haase et al., 2013; Jähnig et al., 2009; Januschke et al., 2009). However, a global metaanalysis on restoration effects was missing as well as a comprehensive, harmonised study considering a broad range of response variables and factors affecting restoration outcomes to get an overall picture on restoration effects and influencing factors (including hydromorphology, different organism groups, as well as food web composition and aquatic land interactions as response variables; and catchment and project characteristics—especially project extent—as influencing factors; 20 case study projects). Here we summarise the results of these studies that were conducted in the REFORM project and place them in the broader context of restoration literature as a basis to derive recommendations for future directions in river restoration.

4.1 General Effect: Does River Restoration Work in General?

Studies reported contrasting and highly variable effects of restoration but there is nevertheless evidence for an overall positive outcome of river restoration. Several studies showed that the ecological effect has been small even if local river morphology and habitat conditions have substantially improved (Jähnig et al., 2010; Lepori et al., 2005; Palmer et al., 2010). In contrast, other studies found a significant positive effect of river restoration on specific organism groups (Lorenz et al., 2012; Schmutz et al., 2014). The few narrative reviews that compiled information on a larger number of restoration projects also found highly variable restoration effects (Roni et al., 2002, 2008). Similarly, variability of restoration effects was high in the few quantitative metaanalyses and a substantial part of the projects showed no effect or even a negative effect, but in general, the overall effect of restoration projects on macroinvertebrates and fish were positive in terms of abundance and/or biodiversity (Miller et al., 2010; Whiteway et al., 2010). These results were supported by the global metaanalysis conducted in REFORM, which found a high variability but an overall positive effect on macroinvertebrates, fish and especially macrophytes (Kail et al., 2015, Fig. 14).

The high variability is likely to reflect real differences in effectiveness of restoration measures applied, as well as context-derived factors such as catchment, river and project characteristics, which either enhance or constrain restoration effect. For example, Kail et al. (2015) reported that nearly half of the variance in restoration effect could be related to differences in project characteristics such as time since the restoration intervention, catchment land use, river size and type, organism groups studied and the biological metric considered as well as the restoration measures applied. The substantial unexplained variance might be partly due to missing information on factors enhancing or constraining restoration effect (Roni et al., 2008) but can also be attributed to noise caused by large methodological differences in respect to monitoring design, field sampling and data analysis. The high variability of restoration effects and our inability to predict restoration outcomes stresses a need for postproject appraisals and adaptive management approaches, which has long been recognised (Downs and Kondolf, 2002; Friberg et al., 1994) but still rarely applied in practise (Williams and Brown, 2014).

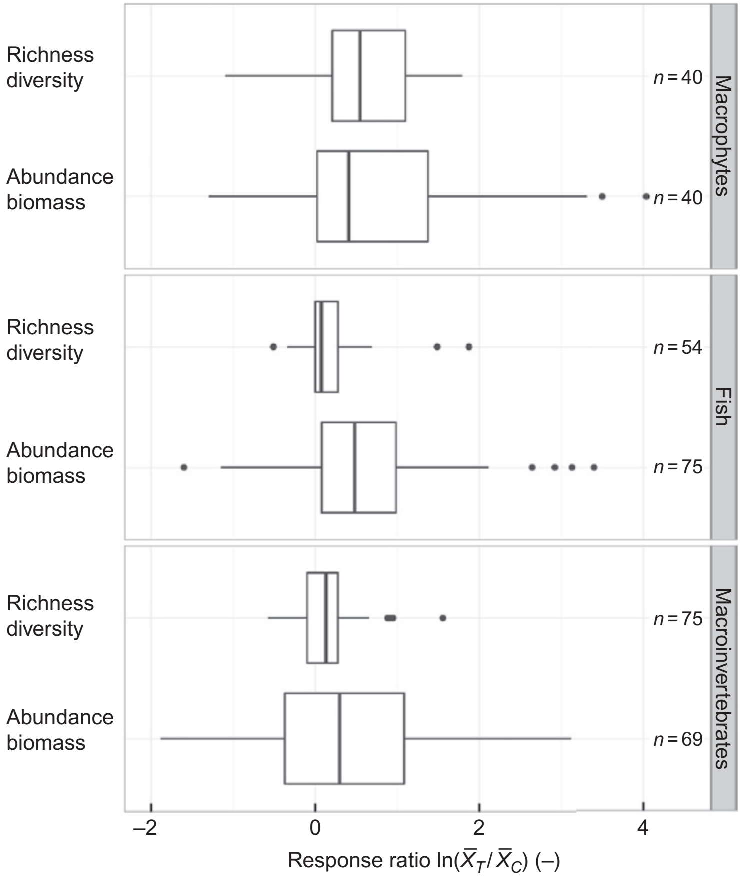


Fig. 14 Overall positive but highly variable effect of restoration on the richness/diversity and abundance/biomass of macrophytes, fish and macroinvertebrates, as reflected by the response ratio of Osenberg et al. (1997) which relates the value of the restored or treatment section X_T to a degraded control section X_C; all mean values were significantly different from zero (t-test, $p < 0.01$). *From Kail, J., Brabec, K., Poppe, M., Januschke, K., 2015. The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: a meta-analysis. Ecol. Indic. 58, 311–321.*

4.2 Differences in Responses Among Organism Groups and Species Traits: Which Benefit Most?

Not all organism groups benefit from restoration to the same extent. For example, studies reported a significant positive effect on floodplain vegetation (Januschke et al., 2011) and macrophytes (Lorenz et al., 2012), a relatively small effect on fish (Schmutz et al., 2014), and a low or missing effect on macroinvertebrate richness and diversity (Friberg et al., 2013;

Jähnig et al., 2010; Palmer et al., 2010). Comparative studies on several organism groups indicated that in general, restoration effect is the highest for terrestrial and semiaquatic groups such as floodplain vegetation and ground beetles, intermediate for macrophytes, lower for fish and the lowest for macroinvertebrates (Haase et al., 2013; Jähnig et al., 2009; Januschke et al., 2009; Kail et al., 2015). Similarly, the effect of restoration significantly differed between the response variables investigated in REFORM as reflected by the Bray–Curtis dissimilarities of the restored and nearby degraded sections (Hering et al., 2015). Moreover, although the effect on richness did not significantly differ between organism groups (one-way ANOVA, $F_{4/89} = 2.082$, $p = 0.09$), there was a tendency for higher effects on terrestrial and semiaquatic compared to aquatic organism groups (Fig. 15).

In the 20 projects investigated in REFORM, restoration had no or only a small effect on species richness or diversity of macroinvertebrates (Verdonschot et al., 2015) and fish (Schmutz et al., 2016), while restoration had a clear positive effect on richness or diversity of organism groups inhabiting river banks or adjacent shallow shoreline habitats, namely ground

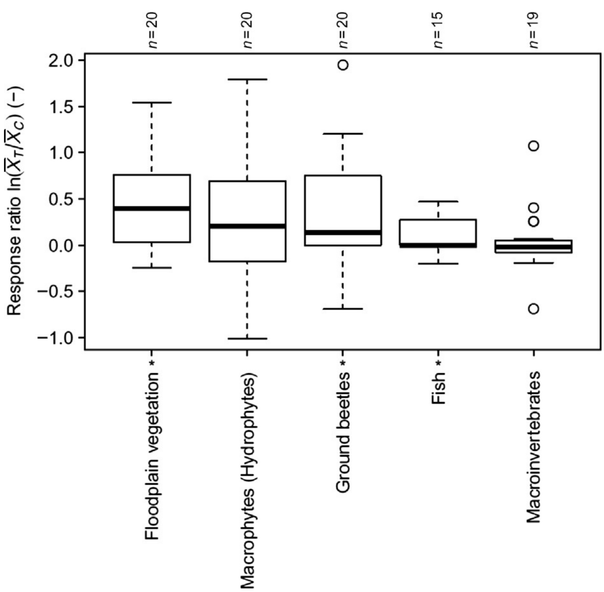


Fig. 15 Effect of restoration on species richness of the five different organism groups investigated in REFORM, as reflected by the response ratio of Osenberg et al. (1997) which relates the value of the restored or treatment section X_T to a degraded control section X_C; mean values significantly different from zero are marked with an asterisk (*t*-test, $p < 0.05$).

beetles (Januschke and Verdonschot, 2016) and macrophytes (Ecke et al., 2016). While Göthe et al. (2016) found no differences in floodplain vegetation diversity between the 20 restored and unrestored sections, restoration had a significant effect on pure richness (Fig. 15), similar to other studies reporting a significant higher richness in restored compared to degraded sections (Jähnig et al., 2009; Januschke et al., 2011). The different effects on different organism groups stress the need to monitor and assess the effect of river restoration on biota in a holistic way, including semiterrestrial and terrestrial organism groups. Terrestrial (floodplain) and aquatic ecosystems are closely linked and cannot be considered and assessed separately, which is clearly substantiated by findings in the REFORM project.

Within single organism groups, restoration generally has a higher effect on the abundance than on the number of species (richness) with Miller et al. (2010) and Kail et al. (2015) finding larger effects of restoration interventions in terms of increased abundance of macroinvertebrates and fish compared with changes in the number of taxa. This is not surprising as taxa that are already present prior to an intervention can respond rapidly with increased population densities to an improvement in quantitative (i.e. more area created) and qualitative (i.e. more variable living conditions) habitat features that allow coexistence of more individuals. In contrast, dispersal of new taxa into a project area is depending on their presence in the system and number of deterministic and stochastic elements (e.g. Leibold et al., 2004). Higher abundance might reflect increase in reproduction and/or the restored reaches attracting individuals from adjacent areas, the latter being especially true for mobile organism groups like fish (e.g. Hughes, 2007). Large-scale pressures like nutrient and fine sediment input suppressing regional to local biodiversity and absence of source populations nearby will limit the possibility of the introduction of new species into a restoration project area.

Our analysis revealed, however, that specific traits are a more sensitive indicator than both abundance and species richness. An alternative method to evaluate the effects of restoration based on species identity (taxonomy) is the use of species traits, an approach recently advocated in relation to biomonitoring (Beche et al., 2006; Charvet et al., 2000; Costello et al., 2015; Demars et al., 2012; Mcgill et al., 2006; Statzner et al., 1997). Traits are inherited characteristics of species that relate to their biology and the environment they live in and provide a mechanistic understanding of how environmental change, such as a restoration intervention, influences biotic communities through environmental filtering (Dolédec et al., 1999; Menezes et al., 2010; Rabeni et al., 2005). For example, richness especially increased for ground beetle species inhabiting sparsely vegetated river banks (Januschke and Verdonschot, 2016), macrophyte richness did only increase for helophytes (Ecke et al., 2016), abundance of small rheophilic fish increased but not for other flow traits (Schmutz et al., 2016), and effects on floodplain vegetation were the highest for metrics describing community structure like the increase of therophytes and annual floodplain vegetation species (Göthe et al., 2016). Moreover, while there was no effect on macroinvertebrate richness, there was an increase of food source diversity as indicated by stable isotopes (Kupilas et al., 2016). These changes in community structure potentially indicate specific functional changes caused by river restoration and should be used in future to increase our understanding how restoration measures affect aquatic ecosystems as suggested for biomonitoring in general (Menezes et al., 2010).

4.3 Differences Between Restoration Measures: What to Do?

Effects of restoration on different organism groups and traits will also depend on the specific measures applied. Comparing effects of different restoration measures in the 20 REFORM case studies indicated that widening (removing bed and bank fixation, flattening of river banks and considerably widening the cross section, and hence increasing the wetted area) is one of the most effective restoration measures, especially for terrestrial and semiterrestrial organism groups (Fig. 16). The effect of widening was more pronounced compared to other restoration measures on ground beetle richness (Januschke and Verdonschot, 2016) and specific macrophyte traits (Ecke et al., 2016). Moreover, the global metaanalysis of peer-reviewed literature conducted in REFORM indicated that the effect of widening is significantly higher on macrophytes compared to fish and macroinvertebrates, but there is also some evidence that the effect of widening on macrophytes may vanish over time (Kail et al., 2015). In contrast, widening had no effect on macroinvertebrates (Verdonschot et al., 2015), which might be due to the fact that—although mesohabitat diversity was significantly increased—widening often failed to increase microhabitat/substrate diversity relevant for macroinvertebrates (Poppe et al., 2015). The high effect of widening on ground beetles and macrophytes might also explain the overall higher effect of restoration on terrestrial and semiaquatic organism groups because these studies mainly investigated widening/rebraiding and remeandering projects. In contrast to planform measures like widening, pure in-stream measures are primarily beneficial for aquatic species with evidence of

Fig. 16 Succession after river widening in a restored section of the river Ruhr (Germany), restored in 2009.

increasing macroinvertebrate and fish richness/diversity (Kail et al., 2015). In accordance, Miller et al. (2010) reported a significant positive effect of typical in-stream measures (large wood and boulder placement) on macroinvertebrate richness.

These differences between restoration measures and organism groups are intuitively meaningful since widening and other planform measures like remeandering reduce flow velocity and create pioneer habitats like bare riparian areas and bare gravel bars that are often sparsely shaded in the beginning, and hence, favour pioneer species in the riparian area and macrophytes in the aquatic zone in the first years. It is only on a longer time scale that natural processes, if these are given room to evolve and substrate can be mobilised, create more diverse aquatic habitats. However, if natural morphodynamic processes are not restored, aquatic habitat quality will not increase and the lack of disturbances to rejuvenate the pioneer habitats will lead to a succession to less favourable conditions for terrestrial and semi-aquatic organism groups as the riparian vegetation develops and shading increases.

This is consistent with the widely held assumption that restoring natural geomorphological processes at relevant (larger) scales, by removing bed and bank fixation and widening, has a higher effect on hydromorphology and biota compared to other nonprocess-based measures like gravel addition into shorter reaches in physically perturbed streams. Therefore, it is crucial to restore a natural flow and sediment regime (i.e. natural processes) and to restore habitat diversity at relevant spatial scales (including microhabitats) to ensure long-term positive effects on all organism groups.

4.4 Confounding Factors: Why Do Some Restoration Projects Fail?

Confounding factors potentially constraining the effect restoration include (i) characteristics of the typically reach-scale projects like the type of measures applied, as well as project size and age and (ii) factors at larger spatial scales like pressures related to catchment land use and missing source populations for recolonisation.

Project size in terms of river length restored might be of importance since longer reaches might mitigate the influence of large-scale pressures like fine sediment input and provide a minimum area for geomorphological processes to evolve and viable populations to establish, and hence, larger effects of restoration should be expected with increasing extent of projects. However, in the 20 REFORM case studies, the effect of restoration did not differ significantly between large and small restoration projects, although there was a tendency for large restoration projects being more successful (Kail et al., 2014). Moreover, the global metaanalysis of peer-reviewed literature conducted in REFORM revealed that restoration had positive effects even though projects were small and effects did not increase with restored reach length (Kail et al., 2015). The restoration projects investigated in REFORM were probably still too small to detect an effect of project size (most case study projects <2 and <2.6 km in the metaanalysis). Other studies including larger projects indeed found that restoration had higher effects in larger projects (e.g. Schmutz et al., 2014). These contrasting results indicate that even small restoration projects can have a positive effect on some organism groups. Slightly larger projects do not necessarily have larger effects (Stoll et al., 2016). Most probably, restoration projects implemented in the past were simply too small to benefit from possible positive mitigating effects of project size and large projects are needed for extensive effects.

Project age might be of importance due to the time needed for natural channel dynamics to create higher habitat diversity and the lag between habitat creation, colonisation and the establishment of vital populations. However, in the 20 REFORM case studies, project age (time between implementation of the measures and monitoring) only had positive effects on aquatic habitat conditions (Poppe et al., 2015) but not on any of the organism groups investigated. Most projects investigated were just implemented 1–16 years prior to monitoring, which was probably less than what is needed for full community recovery (Hering et al., 2015). In contrast, project age was identified as the most important variable affecting restoration success in the metaanalysis of REFORM (Kail et al., 2015). However, the effect of restoration did not simply increase with time but project age had nonlinear and even negative effects on restoration outcome (e.g. on macrophytes). Other studies also report contrasting results on the effect of project age on restoration outcomes (no effect on macrophytes in Lorenz et al. (2012) and invertebrates in Leps et al. (2015), small positive effect on fish in Haase et al. (2013) but different effects in Schmutz et al. (2014) and nonlinear effects in Whiteway et al. (2010)). These results indicate that the effect of restoration does not simply increase over time but changes nonlinearly and might even decrease.

Besides these project characteristics, the effect of reach-scale river restoration is potentially constrained by large-scale pressures related to catchment land use such as water pollution, high nutrient and fine sediment loads, storm water peak flows or a depleted species pool in the catchment resulting in missing source populations for recolonisation. Projects are prone to failure if large-scale pressures are not adequately considered (Bond and Lake, 2003; Miller et al., 2010; Palmer et al., 2010; Roni et al., 2008). In contrast to the numerous studies on the effect of catchment land use on the biological state (Roth et al., 1996; Stephenson and Morin, 2009; Sundermann et al., 2013), there is limited knowledge if and how the pressures related to catchment land use affect restoration outcomes (but see Miller et al. (2010), who found that projects implemented in forested regions tended to have a higher effect on macroinvertebrate richness and abundance). In the REFORM metaanalysis on peer-reviewed literature, agricultural land use was among the three most important factors affecting restoration outcomes for macrophytes, fish and macroinvertebrates, and restoration effect on fish abundance and biomass was significantly negatively related to agricultural land use (Kail et al., 2015). In contrast, there was no clear negative effect of agricultural land use on restoration outcomes in the 20 restoration projects investigated in REFORM. Possibly the reason for this missing clear constraining effect was due to the relatively high share of agricultural land use in the catchments investigated; even the less intensively used catchments might have already been above a critical threshold limiting biota, with any further increase having only a minor impact. Furthermore, at least for macroinvertebrates richness and diversity, the effect of restoration did not depend on the presence of reaches in the vicinity of the restoration projects being in a high or GES, which was used as a proxy for the species pool available for recolonisation. However, effects on macroinvertebrates were nonsignificant anyhow, resulting in a short gradient in the dataset. Therefore, the REFORM results do not question the findings of other studies indicating that restoration effects on macroinvertebrates and fish are limited by the depleted species pool and sparse source populations for recolonisation (Stoll et al., 2013; Tonkin et al., 2014; Winking et al., 2014). This topic clearly merits further investigation since a limited recolonisation potential would need a completely different restoration strategy compared to habitat improvements or pressures related to catchment land use. As a first rule of thumb, source populations for fish and macroinvertebrates should be located less than 5 and 1 km upstream from the restored reach, respectively (Stoll et al., 2013; Tonkin et al., 2014).

In summary, results of the REFORM project, placed in the context of existing restoration literature, indicate that river restoration (i) generally has a positive but highly variable effect, the highest effects are on (ii) terrestrial and semiaquatic organism groups in widening projects, (iii) on aquatic organism groups if in-stream measures are applied, (iv) on species abundance rather than richness and (v) on specific traits rather than mere species number or total abundance. Moreover, results indicate that restoration projects are prone to failure if (i) large-scale pressures related to catchment land use are not adequately considered, (ii) source populations are missing for the recolonisation of the restored habitats, (iii) relevant and limiting habitats have not been restored (e.g. microhabitats for macroinvertebrates) and (iv) related processes like a natural flow regime and sediment transport have not been restored to rejuvenate the habitats. Finally, long-term monitoring is needed to better understand the trajectories of change induced by restoration measures, and to identify sustainable measures which enhance biota in the long-term (Box 2).

5. FUTURE DIRECTIONS

5.1 Future River Restoration Needs Better Planning

The key to improving the effectiveness of future restoration projects is the use of comprehensive and consistent planning, implementation and appraisal techniques. Today, the state of knowledge is primarily based on experiences

BOX 2 Example of River Restoration Project North of the Polar Circle: Unusual Place but Typical in Term of the Trial-and-Error Approach

Lake Børsvann (68.30825N, 16.72100E) was regulated in 1914, and the water was directed away from the river Børselva, reducing the catchment from 85 to 5.5 km^2. By the end of the last century, low discharge and agricultural activities had made the river highly eutrophic, filled with fine sediments and overgrown with plants (Fig. B.1). The water course was protected as a nature reserve in 1997 due to its importance as a feeding and resting area for migratory water birds. A substantial number of different water birds used to nest in the river and lake system, some of them red listed, but there was a drastic reduction in the number and species present during the last years before restoration was initiated. The river was also earlier the main spawning and recruiting area for the lakes downstream, and had a valuable population of arctic char (*Salvelinus alpinus*) and trout (*Salmo trutta*). Fishing with nets and electro fishing showed no fish in 1998 and 1999.

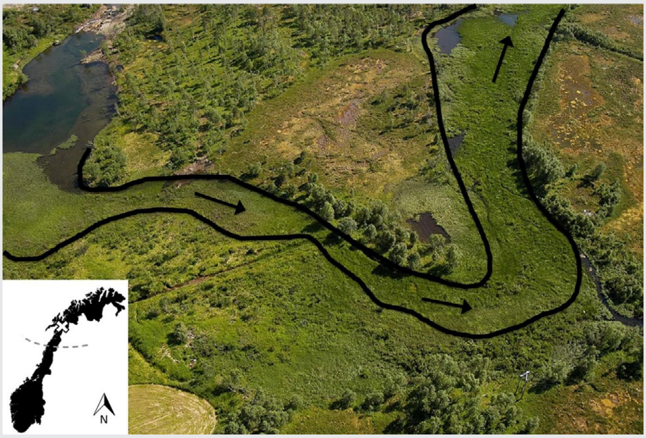


Fig. B.1 The river Børselva with water plants such as *Equisetum fluviatile* and *Carex* spp. covering nearly 70% of the river bed. *Bottom left*: map of Norway showing Børselva (*white cross*) and the arctic circle (*dotted line*).

A 10-year restoration project was initiated in 1997, aiming at: (1) reducing the input of nutrients and fine material from the tributaries. (2) Physically open/alter the river course. (3) Design a new flow regime for the river. Morphology of tributaries was modified to a more natural channel cross section, as they had become very incised by dredging. The more gently sloping banks were covered with coconut mats to prevent erosion until the vegetation was established, and the stream

Continued

BOX 2 Example of River Restoration Project North of the Polar Circle: Unusual Place but Typical in Term of the Trial-and-Error Approach—cont'd

bed was reinforced with geotextiles covered by stones and cobbles to avoid further incision. Furthermore, in the downstream end, a sedimentation pool and two constructed wetlands were created to retain fine sediments and nutrients lost from the fields.

To reestablish the continuum in the system, vegetation had to be cleared to create a new river channel that was dimensioned for the actual discharge regime. Where pools and deeper parts of the river were dug out, an excavator with a long arm was used. The heavy machinery used carpets made for military transport to avoid damaging the natural wetlands (Fig. B.2).

Fig. B.2 Carpets made for military transport over wetlands were used for the excavators to avoid unrepairable damage to the environment, and to create safe passageway for the heavy machinery.

Macrophytes were manually cut, and geotextiles covered with stones were added after clearing, reducing regrowth. During winter, the geotextile and stones were set out on the ice by trucks (Fig. B.3), reducing the need for manpower. This was an easy way of designing bends, side arms and width variation. By varying the thickness of the stone layer and stone size, new habitats for fish and benthic fauna were created. In the end, a new channel was created, connecting the ponds and lakes of the river stretch (Fig. B.4).

BOX 2 Example of River Restoration Project North of the Polar Circle: Unusual Place but Typical in Term of the Trial-and-Error Approach—cont'd

Fig. B.3 Designing a new river with geotextiles and stones during ice cover in winter.

Fig. B.4 Børselva, Norway, post-restoration: a new corridor was created, connecting the different lakes and ponds.

When the new river channel was cleared, the hydrology was restored by changing water release from Lake Børsvatn. This included a minimum residual water discharge and a set number and size of artificial floods in the river Børselva. However, the new water regime was not adjusted to the recreated morphological conditions and caused bank erosion and sedimentation of sand and silt on the newly created geotextile stone habitats. After this, nearly no monitoring or following-up was conducted, and as a result, 2 decades after initiation, the inlet streams were more or less back to prerestoration conditions. Most of the geotextile covered reaches were still open, but there were no published results on substrate structure, flora or fauna in these channels. Furthermore, the river had by itself created new channels, mostly outside the artificial channels.

compiled from a multitude of concepts, tools and techniques used in river restoration over recent decades (Cowx and Welcomme, 1998; Roni and Beechie, 2013). We argue that the way in which knowledge on restoration is available and organised today provides very little concrete guidance to managers wishing to conduct cost-effective projects that deliver set targets. As a part of the REFORM project, we developed an easily applicable project planning framework to support implementation of legislation on surface water by restoration, and to integrate social and economic aspects into decision making. It follows the structure of the well-known project cycle but has been enhanced to assist river restoration in the setting of regional and national objectives by incorporating existing planning strategies (Fig. 17). The framework systematically guides practitioners through two main planning stages of river restoration, from (1) catchment scale to a more (2) project-specific scale, enabling users to put project-specific restoration into a river basin context. It provides detailed information for each of the planning stages and offers tools and guidelines (e.g. Plan–Do–Check–-Act (PDCA), Driver–Pressure–State–Impact–Response (DPSIR) (Angelopoulos et al., 2015), Logical Framework (Cowx et al., 2013),

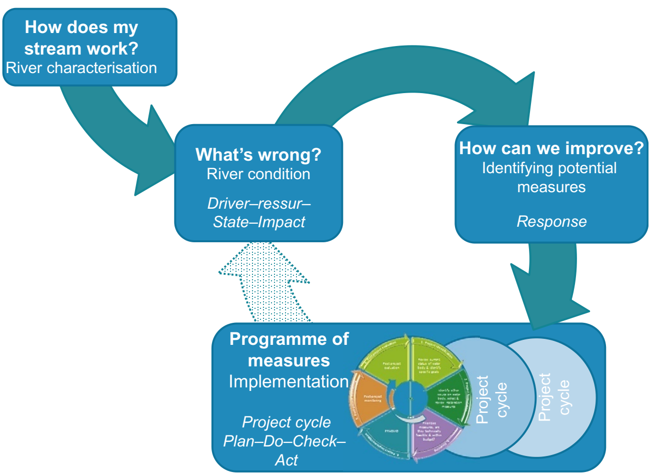


Fig. 17 Project planning cycle at a catchment scale using a five step approach starting at the top left text box: (1) *River characterisation*—at a catchment scale to identify river styles and understand their processes. (2) *River status*—understand the current condition of the aquatic biota or biological quality elements. (3) *River restoration potential*—to understand the level of restoration a river can reach. (4) *Project identification*—to identify specific restoration projects at a reach scale and identify suitable rehabilitation measures and project objectives. (5) *The project cycle*—planning, formulation and implement of projects at a local scale.

Specific Measurable Achievable Realistic Timely (SMART) (Hammond et al., 2011), Before–After Control–Impact (BACI) monitoring, Multi-criteria Decision Analysis (MCDA) (Cowx et al., 2013) and Cost Benefit Analysis (CBA) (Brouwer and Pearce, 2005; Brouwer et al., 2009; Shamier et al., 2013)) to support users (Fig. 18), some of which were developed during the REFORM project. We recommend adopting this integrated project planning framework for river restoration to reduce the uncertainty of management actions by providing five key components:

1. A project planning cycle at a catchment scale, guiding the user through a logical path to design projects linking policy, watershed/catchment assessment, restoration goals, monitoring and evaluation schemes, selection and prioritisation.
2. Concise structured information for each stage of the project cycle, upscaling to river basin to select appropriate restoration measures.
3. Concise structured information for each stage of the project cycle at a project-specific scale and to identify specific measures.
4. Easy access to relevant tools and guidelines that can be used at different stages of the planning process.
5. The choice of more detailed information where needed, giving the option of a more complex planning framework.

These five key components can reduce the uncertainty of management actions and are incorporated in the integrated project planning framework.

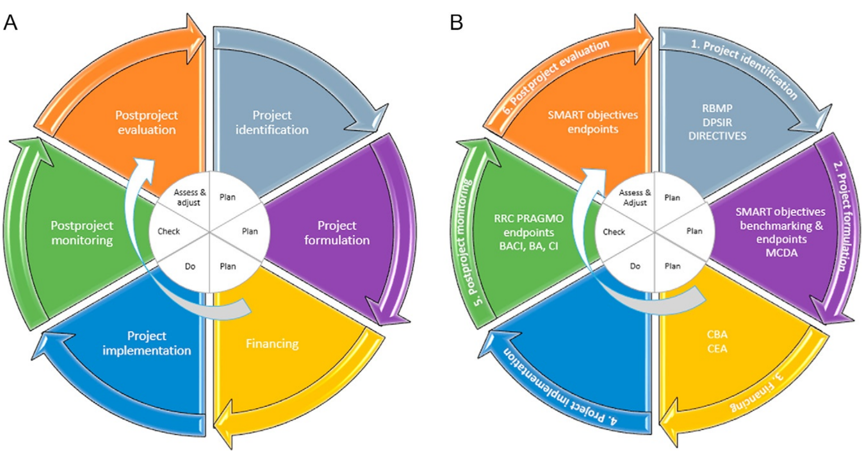


Fig. 18 Project planning cycle at a project scale. (A) Six stages for restoration project planning: (1) project formulation; (2) financing; (3) project implementation; (4) postproject monitoring; (5) postproject evaluation and (6) adjustment or maintenance. (B) The tools needed at each stage.

To aid stakeholders, all information required for the implementation of the framework can be accessed via the interactive REFORM wiki (wiki.reformrivers.eu).

5.2 Project Planning at a Catchment Scale: A Necessity

Applying integrated project planning at a catchment scale takes account of physical, chemical and biological aspects of broad-scale processes of rivers and interfaces between connecting ecosystems, such as the natural habitat continuum from upstream catchments to downstream areas and between the river and its surrounding land use. This ensures river restoration objectives are set to improve ecological status at a river basin level while being defined by institutional, regional and national policies. Subsequent decisions for smaller, local-scale river restoration will still benefit on a larger catchment scale. Here we have categorised project planning at a catchment scale into five main steps composed of *characterising the river* at the catchments scale, assessing its *status* in terms of ecological condition, the *potential* for restoration, project *identification* and implementing the *project cycle* (Fig. 17).

5.3 Exploring the Full Potential of River Restoration

Restoration potential has to be considered across the catchment scale to understand the level of ecological improvement that can be attained. Thus, the current status should be evaluated, including all sector activities that have adverse impacts on hydromorphological characteristics necessary to achieve or support GES, especially as they may further limit the level of restoration that can be achieved. Restoration potential can be assessed when all regional and national policy objectives are considered and constraints in meeting policy objectives are identified. Water body goals and specific objectives identified using all directives/legislations both for and against ecological conservation and finally the current status is compared with objectives to identify constraints.

Synergies between cross-sectoral river ecosystem services and ecological requirements maximise multiple benefits between sectors and enhancing opportunities for restoration of rivers in resource limiting situations (Jackson et al., 2016). Nevertheless, technical barriers to integrated cross-sectoral planning are substantial, related partly to limited data and poor understanding of drivers of each sector, as well as indirect effects of actions (Adams et al., 2014). Thus effective management requires the collaboration between disciplines (e.g. hydrologist, fluvial gemomorphologist, ecologist, economist, sociologist and engineering) and improves the interaction with policy makers and the local stakeholder community to distinguish between the social, economic and environmental requirements of the foreseen project (Letcher and Giupponi, 2005). Understanding these links between ecosystem services and ecological outcomes enables successful integrated planning to produce multiple benefits.

Approaches to achieve multiple benefits are now emerging in river restoration and cross-sectoral interactions, and are supported by various policy documents, for example, synergies between flood risk and river water management or between hydropower development and restoration of longitudinal connectivity for migratory fish (CIS, 2007). The project planning cycle identifies key relationships between society and the environment at both a catchment and project scale to identify and formulate options for synergistic restoration. Here multibenefits can be gained by linking the ecosystem approach, ecosystem services and societal benefits that come from these services to further identify restoration potential and aid decisions for restoration measures.

Agricultural and urban land use, flood protection, inland navigation and hydropower support the key demands of society such as food, water, energy, transport and space to live (Angelopoulos et al., 2015) and these are responsible for pressures that cause biological and abiotic state changes (physical and chemical) and further impacts within the river system. Invasive species and climate change are indirect pressures that can also cause changes in river state and combine with pressures from human activities, intensify impacts on the ecosystem. Disentangling these knock-on effects and identifying mitigation response to the impacts on ecosystem services and ecosystem function through the application of river restoration prevents and improves state changes in the environment.

5.4 Project Identification

For the project identification stage clear objectives are set to improve the health of river ecosystems. Chosen restoration measures are identified at a basin level necessary to meet these environmental objectives. Restoration measures are reviewed and made operational through a number of regulators, therefore, it is important that the planning leading up to the selection of restoration measures is clear and developed in association with regulators and other stakeholders. Catchment-scale planning provides information on river characterisation, river condition and restoration potential, all of which underpin the decisions made to select restoration measures as part of the

PoMs. The end result of project identification is to locate and prioritise reach-scale restoration projects, some of which will combine several rehabilitation measures, in an attempt to ensure smaller scale projects work towards a catchment approach. There is however, no universally accepted approach for prioritising rehabilitation actions and habitat protection (Johnson et al., 2003).

5.5 Project Planning at a Local Scale

A well-designed adaptive planning tool reduces the uncertainty of management actions and establishes the purpose for river restoration to ensure project objectives are set to improve ecological status at local scale, while keeping the project in a river basin/catchment context. The project cycle proposed here is a development of the existing project cycle but includes more detailed planning phases that systematically play a key role when planning river restoration projects (Fig. 18A). The framework has been designed to be transferable to individual restoration projects by drawing on commonalities. Within each stage there are several logical steps to be followed and the option of practical tools and guidelines that can be applied at each specific stage (Fig. 18). A feedback loop provides managers with the ability to account for uncertainty through evaluation of outcomes, and facilitate improved understanding of the efficacy of rehabilitation measures (Fig. 18).

5.6 Project Formulation

In the project identification phase, the project planners should be concerned with the suitability and feasibility of the project, in the formulation phase the emphasis shifts to the acceptability of the project and the desired outcomes. Suitable restoration 'goals' and 'objectives' establish an acceptable state for the system to be restored to, ultimately leading to a self-sustaining river ecosystem (Cowx, 1994; England et al., 2007; Kondolf et al., 2006). However, we still have the recurring limitation to identify restoration success or failure and this is attributable to the absence of adequate aims and objectives besides our limited understanding how ecosystems respond to interventions. Setting benchmarks and endpoints that are linked to clearly defined project goals are a valuable method to help determine the measure of success within river rehabilitation because they provide realistic, quantifiable criteria (Anderson et al., 2005; Buijse et al., 2005). Benchmarks are measurable targets for restoring degraded sections of river within the same river or catchment as representative sites with similar characteristics that have the required ecological status and are relatively undisturbed. Thus attempts to create conditions unrelated to the original ones at the site of interest are avoided and consequently restoration is more likely to result in long-term success (Choi, 2004; Palmer et al., 2004; Suding et al., 2004; Woolsey et al., 2007). Setting benchmarks draws on the assessment of catchment status and identifies restoration needs before selecting appropriate restoration actions to address those needs. Endpoints are target levels of restoration, whether this is an ecological (to restore a level of function/species), social (delivery of services to society) or physical–chemical (hydrology, hydromorphology and water quality) endpoint and are usually linked closely to project objectives. It is important to recognise what is the minimum acceptable achievable level of restoration and what is the desirable level to have as a target endpoint that is still below the benchmark level. Subsequently, what can be compromised for this desired level, will it be cost, ecosystem services or ecological aspects? Natural in-stream habitats consist of complex multidimensional arrays of hydrological and morphological conditions (hydraulic patterns, substrate composition and presence of woody debris) along with the complex life structures and habitat guilds of the biota (Statzner et al., 1988; Strange, 1999) and the environmental conditions (velocity, depth and temperature) and resources (food and space) on which they depend, all of which need to be incorporated in to river rehabilitation. As a result, this level of intricacy needed in river restoration practise is prevented from moving forward as we revisit the reoccurring problem of how to revitalise such a complex systems. The way to move forward is to identify project success of which benchmarking and endpoints will play a vital role in future catchment management.

It is imperative that endpoints accompany benchmarking in the planning process to guarantee the prospect of measuring success because endpoints are feasible targets for river restoration (Buijse et al., 2005). Given that benchmark conditions cannot always be achieved, especially on urban rivers, endpoints will assist in moving restoration effort towards benchmark standards through application of the SMART approach to decide what is specific, measurable, achievable, feasible and realistic in a specific timeframe. There is a need to distinguish endpoints for individual measures, combination of measures, catchment water body measures and river basin district measures. There is thus a need to consider not only the procedures for defining benchmarking and endpoints at the project level but also to integrate the outcomes.

5.7 Monitoring, Evaluation and Project Success

While there is a steady increase of restoration projects each year, the absence of adequate monitoring and evaluation is more frequently a consequence of lack of resources than unwillingness, and this constrains the ability to assess the effectiveness of restoration techniques (FAO, 2008). Monitoring and evaluation plays a key role within the planning framework because it enables identification of river restoration project success by assessing results (outcomes) against objectives (Hammond et al., 2011). Without such analysis it is difficult to assess to what extent the restoration is successful (Possingham, 2012). It is a vital stage in adaptive management as it influences the decisions made to continue, modify or discontinue management actions (Bash and Ryan, 2002). Although the need for monitoring has been acknowledged in recent years (Roni and Beechie, 2013), the majority of river rehabilitation schemes fail to assess outcomes and effectiveness (Cowx et al., 2013). A variety of monitoring techniques are available for detecting environmental impacts of restoration projects whose data collection methods differ spatially and temporally. These monitoring assessment techniques include before/after (BA) contrasts at a single site, BACI sampling sites and repeated BACI and posttreatment design and are well documented in the literature (Ellis and Schneider, 1997; Roni and Beechie, 2013; Sedgwick, 2006). The evaluation phase, for a rehabilitation project which has undergone the initial stages of the project approach, assesses the overall project effects (intentional and unintentional) and the sectoral impact of the project. Evaluation is only possible where a series of measurable indicators or endpoints has been established for the project. The evaluation phase will use measurable indications (in Europe usually WFD compliant BQEs even when not a suitable option as described earlier) to gauge how far the restoration project has developed in relation to the initial objectives and defined endpoints. It should be noted these monitoring tools do not include any CBA, which should be carried out separately.

5.8 Adjustment and Maintenance

Projects may require adjustment when evaluation has demonstrated that the objectives are not reached, i.e. adaptive management. Many regulated rivers have lost much of their natural dynamics lacking rejuvenation of habitats and quite often it is such habitats that improvement projects aim to restore. The altered and regulated conditions, however, may cause such projects to deteriorate and lose their effectiveness, e.g. blocked fish passages by debris or vegetation, pioneer habitats such a bare gravel bar where vegetation settlement has reduced its functionality. They then require maintenance or interventions and to be reset to the initial stage.

5.9 The Future: Holistic and Process-Oriented Restoration

Process-oriented restoration focuses on restoring critical drivers such as hydrological regime and river functions. Process-oriented actions will help to avoid common pitfalls of engineered solutions, such as the creation of localised habitats that cannot be sustained by natural processes (Beechie et al., 2010; Palmer et al., 2014; Palmer and Ruhl, 2015). Restoring natural processes in longer river reaches by letting erosion–sedimentation processes occur by removal of bed and bank fixation; by reprofiling the channel and by reinstalling free water flow has a higher effect on recovery compared to local-scale interventions, such as wood or gravel addition. Overall, the need for a more holistic and process-oriented approach can be condensed into four basic principles in future to make restoration more efficient (Table 5).

Natural flow regimes are the most important driver when restoring rivers (Lorenz et al., 2016) and the success of habitat restoration is likely to be high in rivers with natural flow dynamics, whereas there is a greater likelihood of failure when impaired flow dynamics and processes at larger scales are not addressed (e.g. Jähnig et al., 2010; Palmer et al., 2010). Hydrological measures should therefore focus (1) on groundwater balances and flows at catchment level and (2) on mapping catchment-scale hydrological surface water infrastructure and its functioning. Restoration should involve upscaling of current hydrological measures to reduce discharge dynamics and increase water retention (Richardson et al., 2011; Fig. 19). Other relevant processes such as vegetation encroachment and sediment entrainment are closely linked to water flow.

Holistic measures at the catchment scale and floodplain will have significant hydrological effects also at smaller scales such as individual river reaches. Furthermore, uninterrupted flow and thus lateral connectivity provide continuous potential of exchange of water, substances and propagules. The catchment level is also the appropriate scale to tackle other pressures such as nutrient, organic and toxic load in concert with the hydrological restoration. However, catchment-scale reductions in nutrient emissions are unrealistic in most cases and local impacts of nutrients, organic and toxic substances and sediments can be mitigated at the reach scale by introducing wider or smaller riparian buffers (Figs 20 and 21). There is clear and, in many cases, strong evidence for the role of wooded riparian buffers in controlling nutrient and sediment retention (Hines and Hershey, 2011), water temperature (Kristensen et al., 2013) and improving in-channel habitat structure (Kail et al., 2007).

Small-scale restoration measures should not be ignored in future and profile adaptations at local scales will be necessary in many cases, in particular where stream power is low and there is not sufficient energy to change modified channel profiles back to something more natural (Kondolf et al., 2001;

Table 5 Four Basic Principles to Make Restoration More Efficient in Future

Principle	What to Consider
Target the root causes of river ecosystem change and do this at different scales	Restoration actions that target root causes of degradation rely on knowledge of (1) the processes that drive river ecosystem conditions and (2) effects of human-induced alterations onto those driving processes. This implies that restoration of natural processes with the natural, or near-natural, hydrological, hydromorphological and chemical conditions will have the highest success. In short it means a call for large scale, longer time process-oriented restoration
Tailor restoration measures to the river ecosystem potential in a hierarchical manner	Each river ecosystem is part of a large catchment and the river itself depends strongly on the range of channel and riparian conditions. Both catchment and riparian valley should be or become the logical outcome of the physiographic and climatic setting. Furthermore, understanding the processes controlling restoration outcomes helps to design restoration measures that redirect river valley, river channel and river habitat conditions
Match the scale of restoration to the scale of the problem	When disrupted processes causing degradation are at the reach scale (e.g. channel modification, levees, removal of riparian vegetation), restoration actions at individual reaches can effectively address root causes. When causes of degradation are at the catchment scale (e.g. increased runoff due to impervious surfaces, increased eutrophication), restoration actions need to be taken at catchment scale to restore the root causes
Be explicit about expected outcomes	Process-oriented restoration is a long-term endeavour, and there are often long lag times between implementation and recovery. Ecosystem features will also continuously change through natural dynamics, and biota may not improve dramatically with any single individual action. Hence, quantifying the restoration outcome is critical to setting appropriate expectations for river restoration

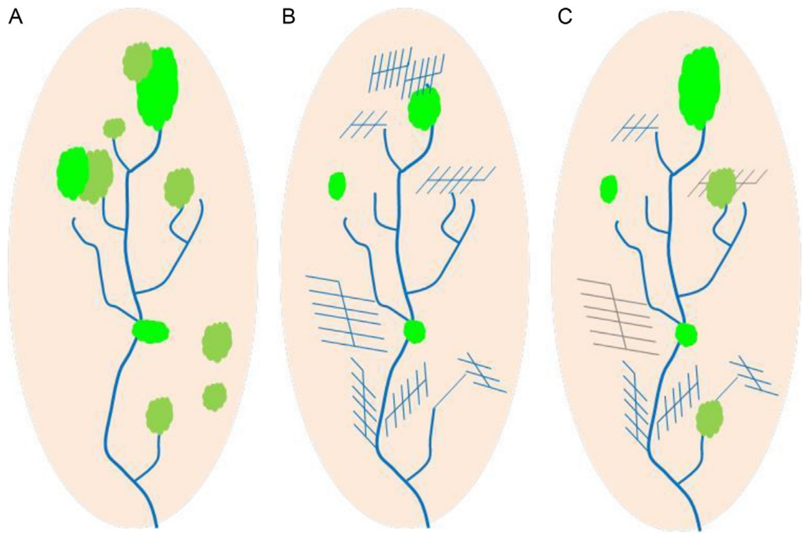


Fig. 19 Hydrological restoration of combined ground water and surface water flows by restoring infiltration capacity and recreating water storage areas at the catchment scale. The image (A) represents a natural catchment with wetlands, bogs and mires that acts as sponges for the water and have a high storage capacity. (B) Catchment degraded by human intervention and a high drainage intensity whereby the natural water stores have been substantially reduced. (C) Restored catchment with water infiltration, reduced drainage intensity, water storage areas (*green* (*dark grey* in the print version)) and water flow retarding by remeandering.

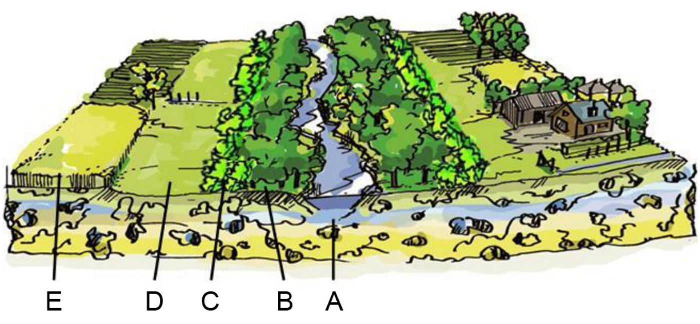


Fig. 20 Installing extended buffer zones at local scale as a restoration measure will control nutrient and sediment run off, cool water temperature and improve in-channel habitat structure (A: river, B: Tree-zone, C: Bush-zone, D: grass-zone, E: adjacent land).

Rinaldi et al., 2016). At local-scale morphological processes (e.g. sorting of bed material, creation of pools, bars and cut-banks) are generally the result of high flows in rich structured beds. By addition of wood or gravel habitat morphology can be improved. Channel incision can alternatively be

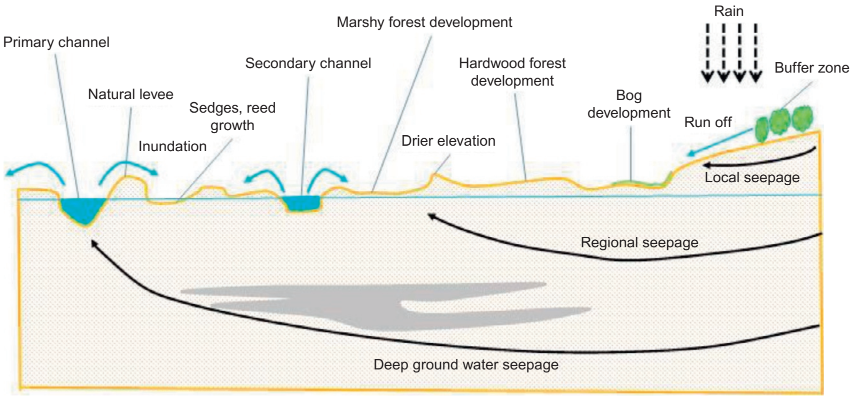


Fig. 21 River reach-scale processes for river floodplain development with natural ground water, rain water and surface water flows that provide the basis for morphological processes. Buffer zone and measures to reduce nutrient levels provide the basis for river restoration.

Fig. 22 The application of large woody debris in combination with the addition of sand in a bare sandy streambed in the Hierden stream (The Netherlands) (*photo left*) results in a strong increase in habitat heterogeneity (*photo right*). *Photos: Ralf Verdonschot.*

decreased or even reversed by placing a large number of naturally shaped logs randomly in the river. Logs in combination with sand addition to the river will heighten the river bed, increase the flow and flow variability and improve habitat heterogeneity (Fig. 22). Changing channel maintenance, such as dredging, removal of wood and weed cutting, from a negative disturbance to a positive measure is another way of improving local hydro-morphological conditions and aid restoration. Soft renaturalisation solutions that are low maintenance and plants, depending on local diversity of growth forms and flow conditions, are bioengineering solutions that create more varied flow and habitat conditions at the river stretch scale (Gurnell, 2014; Iversen et al., 1993).

Table 6 Hierarchical Order of Restoration, Stressors vs Ecological Key Factors and Processes

		Ecological key factors/processes										
	Stressors	***Order of restoration***	***Temperature** regime*	***Light** regime*	***Flow** regime*	***Substrate** variation*	***Organic matter***	***Oxygen** regime*	***Nutrients***	***Salinity***	***Toxicity***	***Connectivity***
Catchment	Changed hydrology	1	■		■	■						
	Diffuse sources	2		■	■	■	■	■	■	■	■	
	Point sources	3	■	■	■	■	■	■	■	■	■	
Stretch	Current alterations	4	■		■	■		■				
	Channelisation	5	■	■	■	■	■	■				■
	Bank degradation	6	■	■	■	■	■	■	■			■
	Maintenance	7		■	■	■	■	■				
Site	Barriers	8	■		■	■		■				■
	Habitat degradation	9	■		■	■	■					

To summarise, one can hierarchically order in nine steps the measures to restore rivers keeping both stress and key ecological processes into account (Table 6).

6. CONCLUSIONS

Results of the REFORM project, and in the context of existing restoration literature, clearly conclude that river restorations conducted until now has had highly variable effects with, on balance, more positives than negatives. The largest positive effects have interestingly been to terrestrial and semiaquatic organism groups in widening projects, while positive effects on truly aquatic organisms groups are only seen when in-stream measures are applied. The positive responses of biota are primarily to abundance of organisms with very little indication that overall biodiversity has increased: specific traits rather than mere species number or total abundance have benefited from restoration interventions. This modest success rate can partly be attributed to the fact that the catchment filter is largely ignored with large-scale pressures related to catchment land use not adequately considered or the lack of source populations for the recolonisation of the restored habitats. The key reason for this shortfall is a lack of clear objective setting and planning process. Furthermore, we suggest that there has been a focus on form rather than processes and functioning in river restorations to date, which has truncated the evolution of geomorphic features and any dynamic interaction with biota. Finally, monitoring of restoration outcomes is still rarely done and often using inadequate statistical designs and inappropriate biological methods which hamper our ability to detect changes.

We suggest that the first step in any future (river) restoration actions is to acknowledge that they are social constructs made by us to improve a wide variety of aspects relating to the riverine environment (Fig. 23). A range of socioeconomic drivers will determine why restoration is considered an option in the first place and what is feasible in the context of potentially conflicting interests. The next steps are then to implement the full restoration project cycle and optimise the restoration effort by using process-based techniques at appropriate scales to maximise ecosystem resilience and recovery.

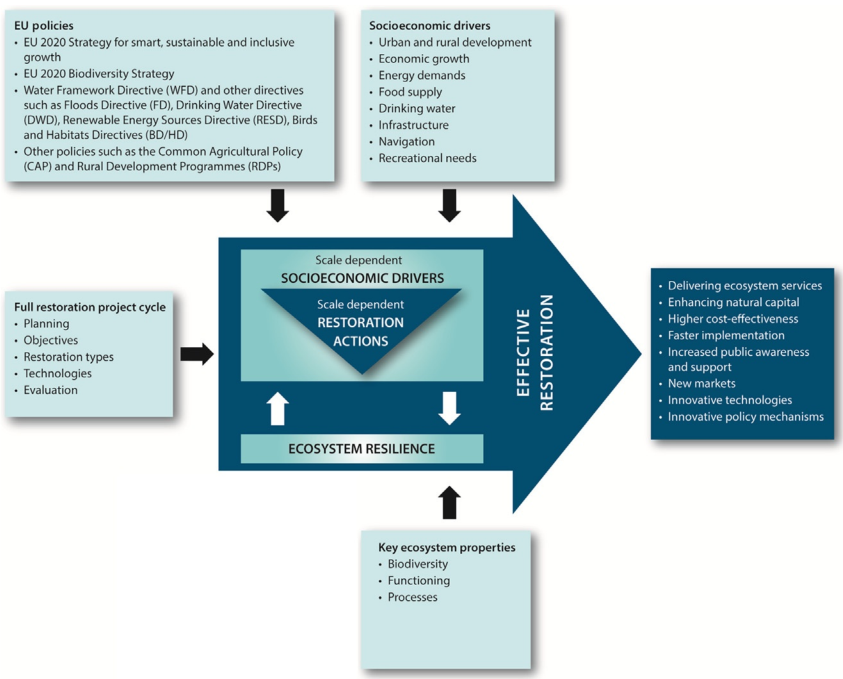


Fig. 23 Effective restoration can only be achieved by considering the societal context in which river restoration projects are set.

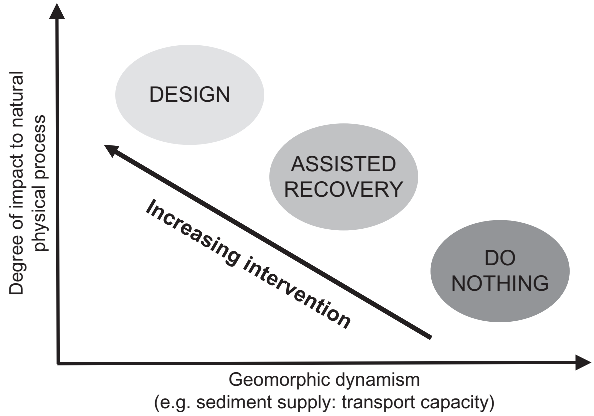


Fig. 24 The need for restoration intervention is a function of degree of impact and geomorphic dynamism of the system at relevant scales.

We advocate the use of a process-based restoration approach based on an initial hydrological and geomorphic assessment of the fluvial conditions across the relevant scales (Fig. 24). Many systems have the potential to recover if they are left to themselves. This is the case in high energy systems with natural geomorphic processes occurring and in rivers that are slightly physically impaired. In contrast, low energy systems with a lack of natural geomorphic processes will need active restoration designs, similar in many cases to what has been done in the past 3 decades. This can be the only option to create a more natural river environment in urban areas or rivers severely impacted by hydropower. In between is the assisted recovery which is exemplified in Box 3. These types of streams have moderate geomorphic dynamism as well as being moderately impacted and are ideal places for soft-engineering approaches involving large woody debris, riparian vegetation and in-stream macrophytes.

The most important catchment-scale stressors that potentially constrain the effects of intermediate and local restoration projects are hydrological changes, disturbed sediments, water pollution and oxygen depletion. The effect of local in-stream and planform measures can potentially be improved by (i) ensuring that catchment-scale pressures do not constrain the effects, (ii) restoring natural sediment dynamics, i.e. processes and (iii) the restored channel pattern corresponds to the channel planform which would develop naturally given the (altered) controls like discharge, sediment load and bank stability. In general, restoring processes should be favoured over restoring form since the risk for failure (created forms being destroyed by channel dynamics) is high.

Better planning acknowledging the social context of all river restoration projects is an essential component in setting clear and realistic objectives (Figs 23 and 25). In future, integration of (river) restoration into the relevant policy agendas could be a strong driver for developing projects and improve excepted outcomes in relation to existing legislation such as WFD and

BOX 3 The Allt Lorgy Restoration in Scotland: An Example of a Process-Based Restoration Approach

Fig. B.5 Assisted recovery by the introduction of wood in the Allt Lorgy changed channel configuration markedly over a 3-year period from a straight single channel to multiple channels with gravel bars.

The Allt Lorgy is a tributary of the River Dulnain in the Cairngorm Mountains, Scotland and part of the Spey catchment. It is a highly hydrologically responsive upland gravel-bed stream that is typical of this region but which was physically managed for agricultural purposes in the late 1980s. This involved significant straightening/canalisation, dredging of the active channel, the construction of artificial embankments (i.e. from the dredged material) and bank revetment/protection, together with in-channel boulder placement grade control structures. The project aimed to restore the physical and ecological conditions of a ~1 km section of the river and its adjoining floodplain and represented a near unique application of the 'process restoration' philosophy (Beechie et al., 2010). This approach aims to reestablish the fundamental physical processes that drive the evolution of a diverse morphology, with the associated benefits for in-stream, riparian and floodplain ecology. Rather than designing a specific channel configuration, the implemented approach of 'assisted recovery' aimed to 'kick-start' natural processes to move the river towards a state of increased physical heterogeneity and improved channel–floodplain connection.

BOX 3 The Allt Lorgy Restoration in Scotland: An Example of a Process-Based Restoration Approach—cont'd

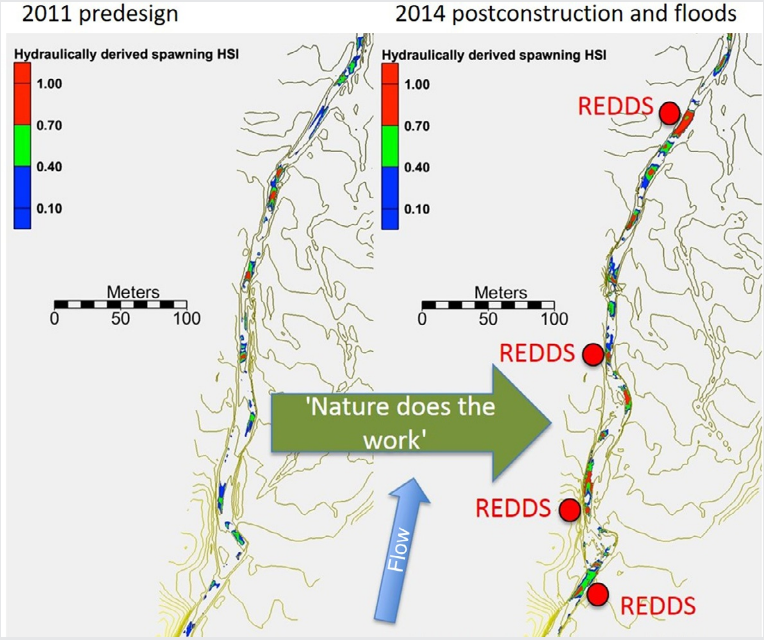


Fig. B.6 Detailed 2D hydraulic modelling of the pre- and postrestoration condition of the site showed a significant increase in hydraulic hetereogeneity. Habitat modelling demonstrated a significant increase in the area of conditions suitable for spawning Atlantic salmon which was subsequently validated by field observations of active REDDS (*filled dots* superimposed on the modelled output map).

The restoration was undertaken in Autumn 2012 and involved a ~1 km section of the watercourse over an 11 ha site. Five stretches of artificial embankments were removed/lowered, liberating over 900 m^3 of material. In-stream boulders were removed and replaced with large wood structures (whole trees with branches and root balls intact). Additional wood structures were introduced in locations identified as key to induce the desired morphological adjustment (e.g. at sites where some degree of meander development had already been initiated). A significant flood event occurred in August 2014, resulting in significant sediment transport, bank erosion and associated morphological adjustment of the channel corridor. A further minor phase of works was conducted in September 2015 with some additional gravel augmentation and minor channel reprofiling.

Continued

BOX 3 The Allt Lorgy Restoration in Scotland: An Example of a Process-Based Restoration Approach—cont'd

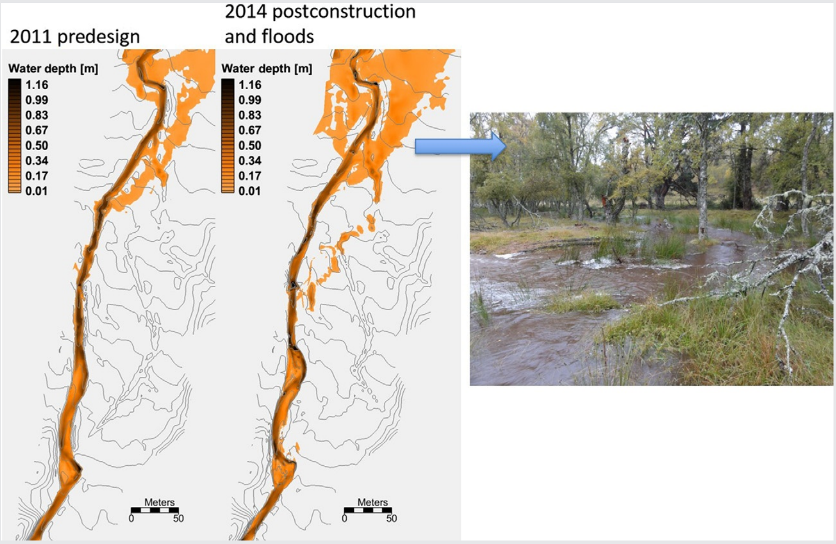


Fig. B.7 Modelling output showing that the restored Allt Lorgy is now better connected to its floodplain, particularly evident in the downstream section of the site.

The site has been extensively monitored to capture physical and ecological/biological evolutions, including repeated high resolution topographic surveys, aerial 'drone' photographic surveys, sediment sampling, REDD mapping, water temperature, electrofishing and invertebrate sampling. Immediately before and after construction, detailed topographic surveys were carried out, with further resurveys undertaken after significant flow events (i.e. 'bank-full' or higher). The monitoring has determined that the restoration measures implemented have significantly reinstated the natural physical and ecological processes that the site would have exhibited under unimpacted (i.e. 'reference') conditions (i.e. an upland 'wandering' gravel-bed river). Three years after design implementation (and a number of high flow events), significant changes in river planform, bedform morphology and sedimentology have been observed. The channel through the site, that had been in a near static 'canalised' condition for ~30 years, has experienced rapid development of three meander bends (~15 m lateral migration, ~2 channel widths) since the restoration measures were implemented. This process has been associated with significant increased sediment storage through the development of complex gravel bar features (i.e. important habitat features), natural tree 'capture' and reconnection of the river with its floodplain (Fig. B.5).

BOX 3 The Allt Lorgy Restoration in Scotland: An Example of a Process-Based Restoration Approach—cont'd

Detailed 2D hydraulic modelling of the pre- and postrestoration conditions of the site showed a significant increase in hydraulic hetereogeneity. This change from the flume-like prerestoration condition greatly improved habitat diversity through the site and there has been a measureable ecological response. For instance, habitat modelling demonstrated a significant increase in the area of conditions suitable for spawning Atlantic salmon, spatially validated by subsequent field-observed REDD distributions (Fig. B.6). The hydraulic modelling also demonstrated greater in-channel shear stress heterogeneity postrestoration, with resultant differential patterns of erosion, transport and deposition of sediments being responsible for the greater physical diversity observed. This has provided enhanced habitat opportunity for a wide range of in-stream and riparian species. Furthermore, the modelling showed that the restored Allt Lorgy is now better connected to its floodplain, particularly evident in the downstream section of the site (Fig. B.7) where the development of a new semipermanent wet woodland habitat has been a direct consequence of the restoration work.

Fish populations are a good indicator of habitat condition and initial monitoring results show significant increases in spawning and juvenile salmon and trout. This provides preliminary evidence that the ecological condition of the channel is responding to physical improvements, although longer-term datasets are required for clear relationships (and the nature of such relationships) to be determined.

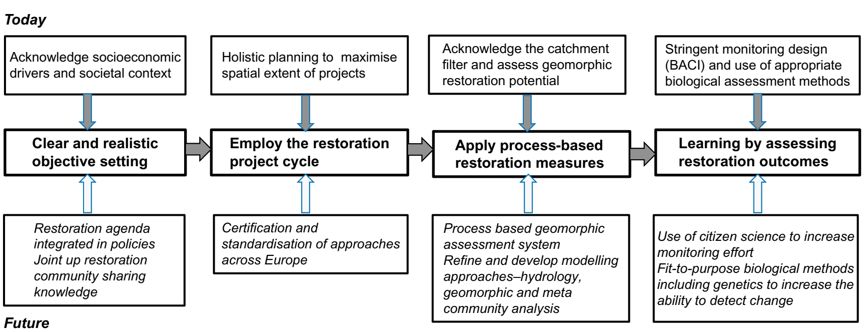


Fig. 25 Summary of the future directions of river restoration-based today's knowledge and future needs for improvement. The future directions can be summarised in four steps that apply to all restorations projects (*central boxes*). *Top boxes* show key elements to be considered for each step from the knowledge we have today. *Lower boxes* illustrate elements that in future need to be implemented to lift river restoration science and application to the next level.

Floods Directive in Europe. Moreover, present knowledge on restoration is scattered among a number actors including policymakers, stakeholders and scientists. A future aspiration should be to promote knowledge sharing by establishing formalised restoration communities. All evidence gathered in the REFORM project, and in the restoration literature, point in the same direction, namely that larger scale projects are more likely to be successful and this take planning (Figs 17 and 25). In future, we need more prescriptive standardised protocols of how to do river restoration that can be certified in a similar manner to analytical approaches. Already today there is sufficient evidence that process-based restoration is likely to deliver better results than restoring solely form (Figs 24 and 25). Hydromorphological assessment methods that record indicators of process and modelling approaches that integrates hydrological, geomorphic and biotic responses are clearly needed in future to aid the development of innovative restoration techniques. Lastly, we need to stop with the current trial-and-error approach to restoration and build an evidence based on how to undertake projects that will deliver the expected outcomes (Fig. 25). We can only do this by monitoring projects in ways in which we can detect change with the necessary degree of certainty. Citizen science and new cost-effective genetic techniques could be some key elements for overcoming the economic barrier than currently hamper a consistent assessment of restoration interventions. The future of river restoration is as bright as we want it to be.

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