This is a draft of a chapter/article that has been accepted for publication by Oxford University Press in the forthcoming book [Freshwater Ecology and Conservation: Approaches and Techniques] edited by (Jocelyne M.R. Hughes) due for publication in 2019.

Chapter 18 - Restoration of freshwaters: Principles and Practice

Carl Sayer[1], Helen Bennion[1], Angela Gurnell[2], Emma Goodyer[3], Donovan Kotze[4], Richard Lindsay[5]

1. **Pond Restoration Research Group, Environmental Change Research Centre, Department of Geography, University College London, UK**
2. **School of Geography, Queen Mary University of London, UK**
3. **IUCN UK National Committee Peatland Programme, UK**
4. **Centre for Water Resources Research, University of Kwazulu-Natal, South Africa**
5. **Sustainability Research Institute, University of East London, UK**

1.0. Introduction

On a global scale, and partly due to our huge dependence on them, freshwater ecosystems are both highly threatened and in major decline (Dudgeon et al 2006). As a consequence and for our own sake in terms of human health and well-being, as well as for the fate of species which reside in freshwater habitats, we need to engage with ecosystem protection and restoration. Conservationists and scientists involved with the restoration of freshwaters face many challenges, including the competing demands of industry, agriculture and water supply, issues of cost, support by the public and government and often the large geographical scale of environmental problems and their causes. Further, it is probably fair to say that restoration practitioners lack a sure-footed, well communicated and widely applicable science basis from which to design and implement successful projects. Nevertheless, although freshwater restoration science is a relatively new discipline, a substantial literature now exists such that some key principles that might underlie successful restoration can be broadly established. This chapter will outline some of these principles and is largely inspired by the classic restoration ecology paper of Bradshaw (1996) who introduces some "*Underlying Principles of Restoration*". Bradshaw's restoration principles include the use of natural processes where possible, recognition that return to an original state may not be achievable but that ecosystem development should be on "an unrestricted upward path", and the need to aim for whole ecosystem restoration which considers both structure and function. Some 20 years on, these principles remain fundamental to freshwater restoration ecology.

The habitats encompassed by this chapter are lakes, ponds, rivers and various wetland systems including fens and bogs. It is not possible to be exhaustive in terms of habitats, environmental stressors and biogeography in a short chapter and hence the subject matter covered herein will be largely drawn from the authors' own research fields in Northern Europe. Issues considered include freshwaters negatively affected by eutrophication and acidification (especially for lakes), habitat degradation and fragmentation (especially for rivers, ponds and other wetlands), habitat conversion to other forms of land-use (especially for wetlands), reductions in water quantity (all habitats) and invasive species (all habitats). Some tropical and dryland case studies from other parts of the world are included, however, and it should also be recognised that the principles and approaches covered in this chapter should be fairly widely applicable and will certainly have relevance to other biomes and aquatic habitats. In this chapter good restoration practice is seen to follow six major principles, which in turn form the basis of key sections and examples. These principles are given as follows:

1. **Restoration targets should recognise the value of historical and pre-disturbance data, but should take into account projected changes in climate, water quantity and other constraining factors;**
2. **Projects need to diagnose the problem and remove those factors that are restraining natural ecosystem re-development and recovery;**
3. **Projects should take account of landscape scale influences and processes that inevitably impact on the success and sustainability of restoration outcomes;**
4. **Where possible, robust Before-After Control-Impact (BACI) style monitoring should be included in restoration projects to help judge success and to inform future work;**
5. **Where possible, restoration should allow and support the capacity of natural processes to repair degraded freshwater habitats and ecosystems;**
6. **Restoration projects should ideally bring scientists, conservationists and stakeholders together using best practice participatory approaches to setting restoration targets and developing restoration designs.**

1.1 Restoration targets

A fundamental question that needs to be addressed before restoration targets can be considered is what constitutes ‘restoration’. Bradshaw (1996) and Brookes & Shields (1996) define ‘restoration’ as the act of returning an ecosystem to its, pre-disturbance, “natural” state, with ‘rehabilitation’ conceptualised as partial progress towards the original state, which is nonetheless not achieved. In addition, other restoration-type activities are highlighted by this work including ‘replacement’ and ‘creation’ which encompass the development of a resource, property or alternative ecosystem that did not previously exist at a location. It follows, given the constraints (**see section 1.2**) associated with populated, modified and heavily modified freshwater landscapes, that rehabilitation and replacement-creation are often the most widely used and practicable forms of restoration.

The types of targets used in freshwater restoration have been many and varied, with water chemistry, water quantity, individual species, biological assemblages, species diversity, habitat structure, ecosystem functioning and ecosystem services all used individually or in combination with one another. In very heavily-modified situations, restoration goals may relate to aesthetics and minimisation of risks to humans, whereas in other settings, goals related to the overall health of the ecosystem and the species it supports have been more common. In particular, for rivers, the term ‘river restoration’ has embraced several perspectives, often reflecting the development of restoration techniques ahead of targets. Initially, river restoration was strongly focused on the physical structure or morphology of the river channel and its mosaic of physical habitats, but then expanded to consider the morphology of the river’s margins and floodplain together with their morphodynamics. More recently, aspects of biogeochemical and ecological functioning have been included (Palmer et al 2014). By contrast, for turbid, plantless shallow lakes affected by nutrient-enrichment, there has been a more stable aim, with most projects focused on returning clear-water, macrophyte-dominated conditions (Jeppesen et al 2012).

But what needs to be considered when setting a restoration target? Is a target always needed? The advantages of having a target are many, allowing restoration practitioners something to aim towards, thus informing management techniques and demonstrating success or otherwise. Certainly the importance of gaining knowledge about earlier stages in the development of freshwater systems has come to the fore over recent decades with the introduction of water legislation such as the US Clean Water Act (CWA; Barbour et al 2000) and the European Council Water Framework Directive (WFD; European Union 2000). Both of these legal instruments require that assessments of water quality and biological assembly be based on the degree to which present-day conditions deviate from those expected in the absence of significant anthropogenic influence, so-called ‘reference conditions’. Similarly, ecological condition assessment of Ramsar Convention wetlands of international significance, and sites designated under the EC Habitats and Species Directive (European Union 1992), are based on comparisons with a baseline state (Gell et al 2013). Nonetheless, there is no universal definition of what 'reference conditions' actually constitute, with definitions ranging from natural conditions in the absence of humans to those best achieved under the influence of humans (Johnson et al 2010).

To assess pre-disturbance conditions in freshwater habitats, various approaches are available. In lakes, ponds and wetlands that accumulate sediments, reference conditions can be derived using palaeoecology, whereby fossil remains of a range of biological groups (e.g. algae, bryophytes, macrophytes, zooplankton, invertebrates and fish) found in dated core samples are used to indicate former environmental conditions, biological assembly and in turn past mechanisms of ecological change (e.g. Hughes et al 2000; Sayer et al 2010). Further, historical records, sketches, photographs and even paintings can be usefully employed to build up a picture of habitat structure in freshwater ecosystems that can subsequently be utilised as a 'guiding image' for what might be achieved by restoration actions (Willby 2011). For example, Madgwick et al (2011) assembled a whole range of historical and palaeolimnological data for Barton Broad, a shallow lake in eastern England, to illustrate the spatial arrangement of aquatic vegetation and past aspects of plant sociology (**Fig. 1.1**). In turn, this image was incorporated into a major review of lake restoration practices to help target and inspire future restoration action (Phillips et al 2015). Similar work has been undertaken for wetland systems, including floodplain fens and peat bogs, with the accumulation of water-logged semi-decomposed plant material as peat, at least where it is intact and undisturbed, affording an excellent record of site ecohydrological development (Whitehouse et al 2008).

While palaeolimnology undoubtedly has much unrealised and future potential in rivers (e.g. Howard et al 2009), space-for-time substitution is a more frequently utilised approach, with reference sites from the best available river examples often used to guide restoration. If these examples are carefully chosen to ensure that they are drawn from a similar river 'type', such an approach can be applied in relation to river biology (Wright et al 2000), as well as hydrology and geomorphology (Rinaldi et al 2016). However, it is increasingly apparent that human impacts on river flow and sediment delivery-transport regimes in many parts of the world are so significant, that true river restoration is rarely feasible. Consequently it is often more appropriate to help a river adjust its form in a semi-natural way in response to the most natural flow and sediment regime that is achievable given inevitable human pressures (**see section 1.2**).

<INSERT FIG 1.1>

Although it is often not possible to restore a site to its natural, background state, knowledge of this limitation and of past conditions is highly desirable, helping to define some of the options for management whilst dictating achievable limits. Indeed, a historically-derived reference state can act as a baseline against which future restoration targets might be assessed and framed. Nonetheless, it is important that a target is actually achievable. In this respect Bradshaw (1996) contends: "*What is crucial is that the development of the ecosystem should be on an upward path in terms of structure and function, and that no barriers to its long term further development can be envisaged*".

There are, however, numerous restraining factors that limit the restoration potential of freshwaters and hence the achievability of a restoration target. These include factors such as high costs of remediation, insufficient water availability due to abstraction (especially for rivers and wetlands), reduced flood disturbance due to flow regulation, an increase in fine sediment delivery due to agricultural intensification (especially for rivers), extinct or declining populations of former species, dispersal limitation, and the influence of other confounding pressures such as legacy pollutants, nitrogen deposition and importantly climate change. It is now widely recognised that climate change may limit the use of historically-derived restoration targets, as the future status of freshwater ecosystems will differ from the present even under ‘do nothing’ scenarios (Battarbee et al 2012; Gell et al 2013; Verdonschot et al 2013), for example as species become eliminated or migrate towards cooler habitats. While this does not invalidate historical targets, there is a need to re-define the reference state as boundary conditions change and, if necessary, adjust them to increase achievability (Battarbee et al 2005). On this basis, in recent years the concept of ‘shifting baselines’ has gained traction (Duarte et al 2009; Bennion et al 2011; Battarbee et al 2014), conceptualising the fact that reference conditions are not only dynamic, but are subject to directional change. In the case of shallow lakes, ponds and wetlands, for example, it has been recognised that valued examples are often transitional states along a hydroseral pathway. Thus restoration targeting needs to consider the point at which restoration actively switches to maintenance management in order to prevent further successional development (Tansley 1939; Sayer et al 2012).

Many studies have demonstrated that, as a pressure is reduced, recovery does not follow a simple reverse pathway, such that ecosystems fail to return to a state that prevailed prior to impact (Duarte et al 2009; Battarbee et al 2014). This may be partly due to lag or hysteresis effects, but a range of confounding factors have also been implicated. In an analysis of nutrient and climate impacts on seven European lakes, Battarbee et al (2012) attributed limited recovery to continuing eutrophication related to an increase in diffuse nutrient loading and/or internal P recycling, but there was also evidence for a climate change role in offsetting recovery. Based on the findings of this study, a conceptual diagram of past, present and potential future trajectories of European lake systems experiencing nutrient-enrichment and climate change was constructed (**Fig. 1.2**). Similarly, in studies of boreal lake recovery from acidification, declines in the richness of invertebrate assemblages have been observed unrelated to changes in acid deposition and more closely associated with climate-related influences on habitat quality, such as oxygen concentrations and temperature (Stendera & Johnson 2008). Thus, much evidence suggests that confounding factors, and climate change in particular, will increase the restoration challenge. In this respect a more dynamic and open-minded approach to restoration will be required - one that considers a range of approaches to deal with an increasingly uncertain future. Perhaps key considerations here are resilience and flexibility, in that restored ecosystems that are resilient and have sufficient room for movement (in the case of rivers in particular) and natural adjustment to changing conditions will likely perform better (**see section 1.2**). In this respect it is crucial to monitor the outcome of restoration projects (**see section 1.5**) and, where appropriate, to use modelling approaches to predict restoration success under future conditions. For example, in the case of English chalk rivers, dynamic models have been used to simulate hydrology and water quality under a range of climate scenarios, revealing a strong association between cessation of drought periods and release of high nitrate loads into the river system (Whitehead et al 2006). When restoration strategies were explored, models suggested that a combined management approach, involving land use change or reduced fertiliser use, water meadow creation, and atmospheric pollution controls could reduce in-stream nitrate concentrations to those of the pre-1950s even under climate change. With such information to hand, restoration practitioners might be able to set more meaningful and achievable goals. Finally, it needs to be recognised that clearly defined restoration targets are not always useful, with this being especially true in the case of re-wilding style projects (**see section 1.3**) where desired restoration endpoints are deliberately left much more fluid.

<INSERT FIG 1.2>

1.2. Diagnosing and tackling the problem

In order to develop appropriate and sustainable restoration designs it is essential to understand those processes, pressures and interventions influencing restoration site(s), how these have changed and how they are likely to change in the future. In turn, such a knowledge greatly assists the design of restoration schemes, enabling attention to be focused on removal of the key factors causing ecosystem degradation. Put simply, if key restraining factors are not tackled, a restoration project is unlikely to meet with success.

The importance of diagnosing and tackling major background problems in freshwater restoration is clearly emerging from the European shallow lakes literature, where, over the last 2-3 decades, many innovative in-lake restoration techniques, aimed at permanently shifting eutrophic lakes from turbid plant-free to clear macrophyte-dominated conditions, have been trialed and studied. These 'internal' measures include biomanipulation (e.g. removal of planktivorous fish, stocking of piscivorous fish, stocking of non-native mussels) which seeks to engineer clear-water conditions by enhancing rates of filter-feeding on phytoplankton, direct planting of macrophytes both within and without wild-fowl enclosures, and measures directed at affecting a reduction in internal P-loading such as sediment removal by suction dredging and in-lake iron addition (see Jeppesen et al 2012; Phillips et al 2015; Phillips et al 2016; Bakker et al 2016). While there are complexities, exceptions and unanswered questions associated with all of these techniques, an emerging pattern is for a lack of long-term and sustained recovery. Biomanipulation is the most fully studied measure, especially in Denmark, where many parallel, multi-decadal studies suggest that positive lake recovery only occurs where nutrient concentrations have been appropriately reduced (perhaps below 50 μg/L for P) or where fish manipulations are regularly repeated, such that the planktivorous fish stock is permanently held in check (Jeppesen et al 2012). Otherwise, although biomanipulation has frequently been shown to generate clear water conditions and macrophyte occupancy in lakes a few years after fish removal, with the recolonisation of fish, phytoplankton-dominated conditions have typically resumed resulting in plant decline after 5-10 years (Jeppesen et al 2012). As a consequence it is emerging, perhaps unsurprisingly, that the key to sustainable restoration success in shallow, nutrient-enriched lakes is effective external nutrient reduction. Thus, nutrient budgets in combination with catchment walk-over surveys to locate nutrient sources, followed by the introduction of measures to reduce external nutrient influx, are probably essential.

Much evidence from other habitats also suggests that a restoration approach which diagnoses and then tackles key underlying causes of degradation as a priority is more likely to be successful. For fen peatlands and many other wetland habitats, it is clearly critical that restoration addresses water quantity issues, with this especially true in arid and semi-arid regions where inflows generally constitute a major component of hydrological inputs to a wetland relative to direct precipitation. In the Murray Darling Basin (south-eastern Australia), for example, the average time-period between environmentally beneficial floods on the Murray River has approximately doubled as a result of surface water abstraction, thus severely disrupting wetland functioning (CSIRO 2008; Pittock & Finlayson 2011). A considerable challenge for these floodplain wetlands is to restore free-flowing tributaries and attain effective management of water releases in regulated portions of the basin to complement flows from free-flowing sections. If achieved, such measures will also enhance resilience to the additional impacts of climate change (Pittock & Finlayson 2011). For many floodplain wetlands, the alteration of geomorphological processes is a further pressure requiring diagnosis in the development of appropriate restoration designs. This issue is illustrated by the Seekoeivlei wetland, South Africa, where introduction of non-native willow (*Salix* spp.) trees to an historically treeless environment drastically altered the geomorphological dynamics of the wetland system leading to the abandonment of a former channel and rapid head-ward growth of a new channel (McCarthy et al 2010). In this case, knowledge of the geomorphological processes underlying these changes, including rates of change, contributed greatly to the formulation, evaluation and hence the sustainability of long-term intervention options.

Another important source of problems in wetlands of major relevance to the development of appropriate restoration designs is human-induced alterations to disturbance regimes. Domestic livestock have been linked to impacts such as increased soil erosion and a decline of sensitive plant species in many wetlands, and in this respect restoration typically involves excluding the influence of livestock, or substantially reducing their numbers (Ramstead et al 2012). However, many wetlands have evolved under the influence of grazing by large indigenous ungulates and if these species are no longer present then the diversity of native plant species and habitats can decline at the expense of one or a few dominants. In such situations grazing by domestic livestock needs to closely simulate the effect of indigenous grazers, and thus part of restoring an 'over-protected' wetland might, in fact, be to introduce domestic grazers where it is not practical to re-introduce large indigenous grazers (Middleton 2013). Similarly, an important problem that often needs to be addressed in the restoration of fire-dependent wetlands may be anthropogenic exclusion of fire. This is illustrated by the KwaMbonambi, northern KwaZulu-Natal, South Africa where, in 1936, herbaceous vegetation comprised 25% of the landscape, but by 2009 there had been a decline to just 2%. A key factor contributing to this change was suppression of fire by plantation forestry management, leading to colonisation by forest species. The KwaMbonambi wetlands naturally support a rich diversity of fire-dependent herbs and grasses, including the only known wild population of the critically endangered herb *Kniphofia leucocephala*. A priority for continued restoration of these wetlands and their associated species is therefore re-instating a regime of periodic burning (Luvuno et al 2016).

A need for effective problem diagnosis is especially the case for river restoration where determining degradation causes and in turn appropriate restoration goals and methods depend upon understanding changing river processes, forms and human activities that extend beyond restoration site(s) to encompass the upstream, and sometimes downstream, parts of a catchment. A prevalent form of river restoration in the late 20th century, particularly within Europe, was habitat-based, focusing on modifying river channel morphology (widening, narrowing, remeandering), introducing stabilising structures (deflectors, boulders), artificially creating habitats (riffles, pools), and implementing planting schemes to provide habitat for specific species or species groups. An underlying assumption of this approach was that the 'renaturalised', more heterogeneous river habitat would lead to biological improvements (Palmer et al 2010). Nevertheless, despite evidence for positive restoration within floodplain and riparian zones (e.g. floodplain plants and beetles – Kail et al 2015; Friberg et al 2016; **Fig. 1.3**), overall the results of small-scale, habitat restoration projects for in-channel biological communities, have been less than convincing. Indeed, recent meta-studies of European river restoration (e.g. Palmer et al 2010; Jahnig et al 2010) have shown that, despite achieving measurable improvements in physical habitat diversity, evidence for significant, positive effects on in-river biology have been patchy, with this especially true for fishes and invertebrates.

<Insert Fig. 1.3>

The reasons for a lack of in-channel restoration success, and indeed a differential response of river and floodplain systems, may relate to a number of factors, not least methodological variation across studies (e.g. specific techniques, time since restoration), continued poor river water quality, dispersal barriers for river invertebrates and fishes, restricted species pools available for recolonisation and the small spatial scale of the restoration work in relation to catchment size. However, a major cause, of failure, even amongst more recent, less engineered restoration projects, is a lack of understanding of the processes, forms and functions that the river is able to sustain following restoration actions, resulting in a non-optimal restoration design coupled with unrealistic restoration targets.

To fully appreciate the relevant processes that may constrain river restoration success, it is essential to go beyond particular river reaches to embrace larger-scale water, sediment and organism transfer processes and their connectivity between river reaches, floodplains and within catchments (e.g. Lake 2012). These are the key fluvial (river flow, sediment transport, water quality) and biological processes that affect and will continue to affect any reach that is to be restored. It is also crucial to recognise that the character of river systems is continually altering in response to changes in these 'natural' processes in combination with human pressures and interventions affecting both restored reaches and the upstream river that influences them. Consequently an understanding of the historical evolution of a river to its present state is essential to developing an appropriate and sustainable restoration design (Grabowski et al 2014), as is an appreciation of how key controlling factors have changed in the past and may change in the future (e.g. Davies 2010; Perry et al 2015). Through new process-based frameworks for supporting restoration design (Gurnell et al 2016; **Box 1**), river restoration is moving into an era where objectives are being more clearly defined, diagnosis of underlying problems and processes is more robust, and a combination of designing with natural processes and incorporating adaptive restoration management is providing a pathway towards genuine and sustainable improvements in river health.

Box 1. The REFORM project

The REFORM project (<u>RE</u>storing rivers <u>FOR</u> effective catchment <u>M</u>anagement - http://www.reformrivers.eu/home) was funded by the EU's 7th Framework Programme (2011-2015). The central aim of REFORM was to provide a series of tools to help improve the success of river restoration.

The REFORM framework (Gurnell et al 2016) exemplifies recent open-ended, approaches to diagnosing the key hydrogeomorphological factors that influence river form and function. The framework (**Fig. 1.4**; Gurnell et al 2016) has been developed for application in a European context (e.g. England & Gurnell 2016) and helps to diagnose the key factors influencing form and function in river reaches. It considers individual river reaches in the context of the valley and river segment, landscape unit and catchment within which they are located. Further, it incorporates three main stages of analysis: 'delineation' of spatial units; 'characterisation' of contemporary and historical key processes within the spatial units through the generation of indicators; and 'assessment-diagnosis'. Four types of assessment and diagnosis are conducted based on information, particularly the values of the indicators, assembled during the characterisation phase. First, the current form and function of individual reaches is assessed, including their channel sediments, morphology, vegetation and degree of human alteration; the character, function and artificiality of their riparian corridor; and any evidence of current morphological adjustment. Second, past and present indicators of water and sediment production, transfer and delivery from the catchment through the river network are assessed at all spatial scales. Third, reach-scale indicators of historical morphological adjustment are assembled. Finally, the results of the historical and contemporary analyses conducted in the three previous assessments are combined, to summarise space-time changes; to unravel causes and responses in order to understand trajectories of change that have occurred; and to consider likely responses to specific future scenarios such as the effects of climate and management change. Such a multi-scale approach to diagnosing why a river reach has a particular form and dynamics is absolutely essential to designing restoration interventions that 'work with the river' to achieve sustainable improvements.

<Insert Fig. 1.4>

1.3. Good restoration encourages the natural repair of degraded systems

In his classic paper Bradshaw (1996) emphasises the huge advantage of utilising natural recovery processes in ecological restoration, pointing out that, in the North American Great

Lakes region, pre-existing soils and vegetation were repeatedly destroyed by ice ages, but were subsequently able to build up ecosystems of high complexity and diversity. In other words, nature is a powerful force that should be allowed to heal ecosystems wherever possible. It follows, therefore, that good restoration helps to stimulate and enhance natural physical and ecological recovery processes wherever possible. It is probably true that many freshwater habitats, especially rivers and wetlands, are capable of natural repair, with the removal of factors that hamper recovery, but frequently the longer timeframes required for natural recovery have meant that such approaches have often been ignored.

There is mounting scientific evidence for the importance and potential of natural processes in freshwater restoration. For example, in restoring wetland vegetation, two common and opposing restoration practices are self-design vs. intensive revegetation (O'Connell et al 2013). Self-design restores hydrogeomorphology without the artificial introduction of plants (e.g. via seeds and plug-plants) into restoration sites. By contrast, intensive revegetation does the same but involves the often costly and time-consuming inoculation of sites with plants. Self-design has many advantages, being cheaper and potentially resulting in a more natural, local vegetation, but there are risks and potential disadvantages (e.g. invasive species arrival, soil erosion) if vegetation fails to colonise rapidly due to a highly depleted seed bank and/or limited existing plants on site (Weinhold & van der Valk 1989). Further factors affecting unassisted recolonisation include the extent to which the native flora and fauna is dispersal-limited and the proximity of intact areas from which colonisation can occur.

Inspiring examples are emerging of cases where restoration based on natural plant recovery has led to the return of surprisingly high biodiversity to freshwater systems. For UK agricultural ponds lost to land consolidation (so-called "Ghost Ponds"), Alderton et al (2017) showed that many aquatic plants can colonise resurrected pond basins from long-lived (150+ years) seed banks. Equally rapid, pond-wide re-colonisation of aquatic vegetation for overgrown farmland ponds restored through scrub and mud removal strongly suggests the same effect (Sayer et al 2012), as illustrated in **Figure 1.5.** A particularly spectacular example of rapid plant re-colonisation following restoration comes from Lake Fil in Denmark, a large shallow (mean depth 1.5 m) lake drained to permit agricultural land expansion over 1852-1952 (Baastrup-Spohr et al 2016). Within just two years of re-establishing the lake it developed a remarkably high diversity of aquatic macrophytes (33 species), many of which were locally and indeed nationally rare. This occurred despite intense farming of the land for 50 years prior to restoration. Similarly, in the case of fen peatlands destroyed by agricultural land development, diverse fen assemblages have been restored within a very short amount of time by stripping off agriculturally enriched uppermost layers to expose seed banks and by allowing plants to self-establish (McBride et al 2011).

<INSERT FIG 1.5>

In river systems, natural vegetation recolonisation of restored sections can also be successful if propagule delivery is efficient from upstream species pools. For example, restoration of a section of the River Cole, UK, involved cutting a new sinuous river channel in its historical location to bypass a realigned section (Gurnell et al 2006). No soil, seed or plants were applied to the newly cut river banks and yet 145 plant taxa were identified within the seed bank and standing vegetation after just two years of the river being diverted into the new channel. Furthermore, the colonising vegetation did not include any widely occurring non-native invasive species commonly found along UK rivers (e.g. Himalayan balsam *Impatiens glandulifera*, Japanese knotweed *Fallopia japonica*) despite a predominantly urban upstream catchment (Gurnell et al 2006). Thus, there are strong arguments for always considering the potential for natural re-vegetation of freshwater habitats in restoration projects, be they rivers, lakes, ponds or wetlands. More research to address this theme is urgently needed in order to identify where and when natural re-revegetation is effective, particularly in relation to factors such as former land-use, hydrological setting, soil type and chemistry and importantly propagule longevity for different plant species (e.g. Stroh et al 2012; Bakker et al 2013).

A particularly promising approach to stimulating natural recovery in freshwater environments is to utilise 're-wilding', whereby natural, often missing processes are re-introduced and/or allowed to operate with minimal human interference. In freshwater environments this idea encompasses a range of examples, including land abandonment and the removal of management (Navarro & Pereira 2012), re-introduction of natural tree-fall into rivers (Thompson et al 2018) and the re-introduction of apex predators (Beschta & Ripple 2016). The most extensively studied and perhaps the most dramatic demonstration of re-wilding benefits to freshwater systems probably comes from the re-introduction of beavers. Both American beaver (*Castor canadensis*) and European beaver (*Castor fiber*) have started to recover in the wild during recent decades assisted by several re-introduction projects. Where they have returned to wetlands many positive changes in both hydrogeomorphology and ecology have been observed. In particular, beaver introduction (and the subsequent natural dispersal of individuals), together with consequent habitat modification via tree-felling, grazing and dam building, has led to an extensification and diversification of wetland habitat with significant biodiversity increases demonstrated across a range of aquatic and semi-aquatic organisms including plants, invertebrates, amphibians, birds and bats (Nummi & Holopainen 2004; Cunningham et al 2007; Law et al 2017; **Fig. 1.6**) as well as increased flood attenuation and reduced diffuse pollution (Puttock et al 2017). Although more studies are needed, especially in terms of long-term responses and dynamics, as suggested by Law et al (2017), in many cases beaver introduction has achieved successes that far out-weigh the benefits of more engineered approaches to freshwater restoration such as pond creation and river re-meandering. The mechanisms that drive beaver-benefits are still to be fully understood, however, as they often involve the re-instatement of hitherto little understood processes and indirect species interactions, including linkages across the aquatic-terrestrial interface (see McCaffrey & Eby 2016). For example, introduction of beavers may help to restore wetland vegetation to former agricultural land not only through lifting the water table but also through the disturbance and exposure of old seed banks (Law et al 2017).

<INSERT FIG 1.6>

A re-wilding approach to freshwater and wetland restoration clearly has considerable potential, but constraints must also be recognised. For example, in the case of peat bogs, natural recovery is probably wholly possible, given the removal of adverse pressures and sufficient time. However, the timescales involved in the natural recovery of full ecosystem diversity and function following extensive peat extraction or severe erosion, for example, may span many centuries or even millennia and are thus often unacceptable in terms of lost ecosystem services over such a long period. Consequently there is generally a strong argument, or at least a strong incentive, for intervention in order to hasten the natural recovery process. In particular the setting of targets and timescales for delivery are common features of funding conditions which tend to conflict with re-wilding ideals, typically resulting in a more hands-on and short-term focus to system restoration actions. In the case of beavers and large predators (e.g. lynx and wolf), human-wildlife conflicts are also a key issue that delay or sometimes prohibit re-wilding attempts (Lorimer et al 2015). Finally, in urban and intensely managed and populated areas, re-wilding approaches to restoration are often precluded due to severe constraints in terms of space and general flexibility. Nonetheless, even where full and natural hydrogeomorphic and ecological processes cannot be set in motion in an ideal way, elements of this ethos can be applied to most settings and one positive aspect of a re-wilding approach is its ability to inspire, encourage and sometimes completely re-energise restoration practitioners.

1.4. Recognise importance of landscape-scale influences and processes

Given the non-local nature of many freshwater environmental problems (e.g. diffuse pollution, catchment-scale barriers to species migration, over-abstraction), there have been growing calls for a restoration approach that considers larger spatial scales and looks outwards from the site-scale to the wider landscape and beyond (as summarised in the context of rivers in **Box 1**). Indeed, it is now fairly widely recognised that conservation and restoration need to extend beyond the boundaries of existing protected areas (Adams et al 2016). To date, however, although there will be many exceptions, most freshwater restoration work has been undertaken on relatively small patches, for example individual river reaches (typically <1 km), floodplains and lakes and relatively small collections of ponds. In particular, where small units of river have been worked on, but where key negative catchment-scale influences have been not been sufficiently tackled, restoration projects have often failed to deliver significant measurably improved outcomes (Palmer et al 2014), probably partly for this reason. For example, where a lowland river, lake or wetland system is severely impacted by excessive nutrient and sediment inputs from an upstream agricultural catchment, local habitat restoration in rivers and within-lake restoration measures are likely to have relatively dampened benefits (**see section 1.2**). Similarly, local river restoration activities directed at fish (e.g. gravel introduction, re-meandering), where barriers that prevent natural upstream-downstream fish migration have not been removed, are less likely to have positive effects on fish assemblages (Champkin et al 2018). The spatial scale of a restoration project in relation to degree of influence from broader catchment-scale pressures is thus critical to restoration success.

There are strong arguments for freshwater restoration projects and strategies that are as ambitious as possible in the spatial scales they encompass and which recognise the importance of larger scale system connections both longitudinally (upstream - downstream) and laterally (links to riparian zones) for fluxes of water, pollutants, key substances (nutrients, sediment, carbon) and movements of propagules and species (Friberg et al 2016; Fergus et al 2017). From a biodiversity conservation perspective much research has shown that the connectivity between different aquatic habitat patches has an important, often positive, influence on biological structure and biodiversity. It follows, therefore, that a more open-minded approach to freshwater restoration, one that considers the full spectrum of aquatic habitat patches in a landscape – an "aquatic landscapes" approach - is likely to be more successful (Sayer 2014). For example, river restoration projects which have facilitated enhancements to floodplain and associated pond and backwater habitats have been shown to have more substantial benefits than restoration of the river channel on its own (Sayer 2014; Friberg et al 2016; **Fig. 1.3**). Equally, by undertaking catchment-wide and strategic studies of barriers to fish passage, it is possible to ensure that restoration by barrier removal work has maximum effect (Perkin et al 2015). In the case of wetlands it is becoming increasingly evident that successful restoration of native wetland vegetation depends, not only on re-creating favourable on-site conditions, but also the location of a wetland in relation to other neighbouring sites. Specifically, a wetland in close proximity to intact areas of native wetland vegetation is generally much better placed in terms of natural re-colonisation by native plants than are distant, isolated wetlands (Findlay & Houlahan 1997; O'Connell et al 2013).

A catchment-scale approach to restoration which looks firstly to address issues of headwater pollution and which aims to facilitate freer movement of water, propagules and species throughout an aquatic network may be the ideal (Sayer 2014), but of course there are a number of caveats. For example, it is not always possible to resolve upstream pollution problems in their entirety due to the predominantly anthropogenic nature of many catchments. In some cases, local scale buffering may be sufficiently effective (Weisstenier et al 2013) but it also needs to be recognised that larger scale problems such as transboundary pollution and climate change cannot be addressed at the individual project scale. Furthermore, removing hydrological barriers can increase the spread of invasive species, so decisions need to be made on the pros and cons of enhancing connectivity, especially where rare native taxa could come under threat. The important point here is that practitioners should at least seek to obtain sufficient knowledge of wider catchment and landscape influences, so that informed decisions about the most appropriate restoration approach can be made.

1.5. Importance of monitoring restoration

Monitoring is essential for providing assessments of the success or otherwise of freshwater restoration, as well as to assist the design of more effective restorations in the future and, importantly, to help make the case for future investment in this field. In order to judge the effectiveness of freshwater restoration activities it is crucially important that data are collected pre- and post-restoration and that they are quantitative; qualitative sampling is incapable of assessing changes to species population size and also makes biodiversity changes harder to assess. It is also important that restoration monitoring of biology incorporates measurement of key hydrogeomorphic and chemical variables, such that the underlying mechanisms which affect restoration success/failure might also be inferred. Finally, it is essential that unrestored 'control' sites are included to account for natural and other background drivers of change. Such an experimental set-up represents the Before-After Control-Impact (BACI) approach. An even more ideal set up providing enhanced statistical power would be to undertake multiple concurrent restorations focused on the same type of intervention (hence MBACI; Thompson et al 2018). However, while the rationale for BACI is strong, it is often the case that restoration projects have neglected, or do not have sufficient resources to incorporate appropriate control sites, making it impossible or at least difficult to arrive at clear conclusions about restoration success (Feld et al 2011).

As well as adopting a BACI approach, a crucial aspect of restoration monitoring is that it is established and funded to cover appropriate time-periods. In a review of river restorations in Europe and North America, Feld et al (2011) revealed that the majority of studies spanned a period of just 1-7 years, a period shorter than might often be required to detect biological recovery. Indeed invertebrate response to river restoration that is limited by poor water quality (Kail et al 2012), can take 5-10 years before any change is detectable, while fish populations often take decades to change in response to perturbations (Trexler 1995). For example, wetland trees such as willow and European alder (*Alnus glutinosa*) may require decades to mature, such that the full function of riparian buffer strips and trees in rivers (temperature modification, large wood recruitment) rivers may take 30-40 years to be achieved. Similarly, although the restoration of peat-forming vegetation may occur relatively quickly as a result of restoration actions such as ditch and gully blocking, the processes involved in establishing full ecosystem diversity and function require more than just a few decades of monitoring.

Due to funding constraints, long-term monitoring is necessarily restricted to a few sites, but the evidence gathered from such projects is incredibly valuable. For example, 20-30 years of monitoring Danish shallow lakes is starkly revealing the shortcomings of in-lake restoration techniques such as biomanipulation (Jeppesen et al 2012). Further, in the UK, 30 years of the Acid Waters Monitoring Network (AWMN; now the Uplands WMN), based on 22 lakes and streams, has been instrumental in assessing the recovery of freshwaters from acidification, revealing that, while marked changes in deposition chemistry and water chemistry are consistent with recovery, the extent of biological recovery remains somewhat limited (Battarbee et al 2014). Given the slow operation of some hydrogeomorphic processes, as well as lags and delays in responses of some long-lived freshwater components and, importantly, confounding and other background influences on freshwater systems, it is evident that short-term 'typical' restoration monitoring studies can only provide limited insight into system responses to restoration. It is therefore crucially important that scientists and indeed the public fight against an ever growing trend of cutting funds for freshwater monitoring stations and networks. In essence, freshwater restoration science will only be as good as the quality and indeed quantity of system monitoring that takes place and without monitoring the evidence on which restoration designs are based will inevitably be weak.

Finally, and linked to the arguments for stakeholder working made in **section 1.6**, it is important that knowledge gained from restoration studies be clearly and sensitively communicated to practitioners and stakeholders involved in restoration work so that the adoption of evidence-based approaches is taken up early and mistakes of the past are not continually re-made.

1.6. Working with stakeholders

Human well-being, both physically and psychologically, is intimately connected with freshwater. Consequently freshwater restoration activities can have long term benefits for humans as well as ecosystems, although this is not always fully appreciated. This means that there is a crucial need to highlight to the public and indeed to governments the vital importance of clean freshwater and of healthy, species-rich, more naturally functioning freshwater systems. Restoration is also likely to be better funded (and for longer perhaps), supported and valued if such societal relevance is made clear. Equally, to achieve restoration success it is crucial that stakeholders are on board with a project's aims and see its importance right from the start while also being fully engaged with the restoration process. There is much evidence to suggest that partnership approaches to restoration which involve the local community and landowners as well as more formal stakeholder groups (e.g. conservation organisations) are most likely to achieve the most sustainable outcomes (e.g. Eden & Tunstall 2006; Åberg & Tapsell 2013).

Full engagement of all stakeholders requires early and effective communication of what is proposed to enable the multiple objectives and potential win-win aspects of restoration to be appreciated by a wide audience. Communication should not be one-way. It needs to proceed in multiple directions in order to integrate stakeholders fully into discussions and decision-making. To achieve this requires the adoption of diverse participatory techniques (e.g. Moran et al 2016) in order to build mutual trust. An increasing emphasis on the many ecosystem services delivered by freshwater systems when they are functioning well (e.g. Acuna et al 2013) can be an effective vehicle for quantifying and thus communicating restoration benefits (e.g. Vermaat et al 2016). Such an approach can extend beyond individual small projects to support an integrated understanding of the potential benefits of multiple combined projects or very large individual restoration schemes. One model which has worked well is the establishment of projects under a branded partnership umbrella with funds secured for core staff to communicate both the value of the work with stakeholders and to manage the restoration works. Having these partnerships provides some longevity to a suite of short-term projects, allows for co-ordination of funding and provides a point of contact for stakeholders and the public. The partnership model of delivery is extensively used in the UK to deliver effective and long-term peatland restoration projects (Cris et al 2011) and for river systems through the establishment of catchment partnerships.

Acknowledging that the general public may have perceptions which differ widely from scientists, and the possibility that memories of the past may not always be accurate due to some level of personal amnesia and shifting baseline syndrome (Papworth et al 2009), local people nonetheless often have valuable knowledge of a site, sometimes accumulated over many generations. This knowledge should be harnessed and can be extremely valuable in guiding the restoration process. Furthermore, in recognising that stakeholders, including local people, may have widely divergent opinions on what constitutes a problem worthy of restoration and what the endpoint should be in addressing restoration problems, there is a need for open discussion of the perceived values of systems in different states in order to facilitate effective and hopefully more consensual decision-making (Hobbs 2016). Such desired open discussions can be difficult to achieve when divergent interests, norms, values and perceptions are brought together, and approaches such as social learning offer useful means of providing people from different backgrounds a 'safe space', to share their experiences and to develop new knowledge, ways of thinking and possibilities (Wals 2007).

Given a frequent need for public participation in freshwater restoration, a degree of compromise is often needed, with this being especially true in highly populated urban settings. For example, public perceptions on what a successful wetland restoration outcome is (typically neat, picturesque and with open water) generally differ greatly from what would be the ecologically functional or reference wetland determined from a scientific basis (Nassauer 2004). Such "ecologically-directed" wetlands often lack open water and appear to the general public as 'untidy'. It is therefore suggested that, even if wetland restoration is being designed primarily to achieve a reference ecological state (whether pristine or functionally-defined), the restoration plan may need to include recognisable and valued landscape characteristics to improve the likelihood of it being sustained through societal support over the long term (Nassauer 2004). It may also be important to take into account that public perceptions of what constitutes a threat requiring restoration/rehabilitation intervention may also differ greatly from that determined through scientific assessment (Schumm 1994).

1.7. Conclusions

Freshwater systems afford some of the most biodiverse and culturally important habitats on the planet and it is essential that we rise to the restoration challenge. While the science of freshwater restoration is developing at a pace, uncertainties about how we repair the hydrogeomorphology and ecology of freshwater habitats remain. More studies and, importantly, high quality monitoring work are therefore required. Many advances have been made with regards to problem diagnosis, restoration targeting and importantly the selection of appropriate restoration approaches. But one thing is clear restoration is easier when the extent of system damage is reduced. Thus, a key lesson is to identify and then protect and conserve existing high quality sites and to enact restoration work before human-induced degradation is too severe. For example, it is much easier to restore a shallow lake that has not already lost its macrophytes. Furthermore, in the case of peat bogs, given the long timescales required for natural peat formation and accumulation, the key human action is not to damage such sites in the first place. We cannot rely on restoration activities to sort out all our problems and often rare and declining species cannot afford to wait until a large-scale ecosystem restoration programme is brought into being. Early intervention is key, and the central messages of Bradshaw's classic paper undoubtedly hold true: good restoration should look to tackle the root cause of the issues that are degrading a freshwater system, but wherever possible we should give freshwater systems the space, time and flexibility needed to repair themselves via natural processes.

References

Åberg & Tapsell (2013). Revisiting the River Skerne: The long-term social benefits of river rehabilitation. *Landscape & Urban Planning*, 113, 94–103.

Acuna, V., Diez, J.R., Flores, L., Meleason, M. & Elosegi, A. (2013). Does it make economic sense to restore rivers for their ecosystem services? *Journal of Applied Ecology*, 50, 988–99

Adams, W.A., Hodge, I.D., Macgregor, N.A. & Sandbrook, L. (2016) Creating restoration landscapes: partnerships in large-scale conservation in the UK. *Ecology & Society*, 21(3), 1.

Alderton, E., Sayer, C.D., Davies, R., Lambert, S.J. & Axmacher, J.C. (2017). Buried alive: Aquatic plants survive in 'ghost ponds' under agricultural fields. *Biological Conservation*, 212, 105–110.

Bakker, E.S., Sarneel, J.M., Gulati, R.D., Liu, Z. & van Donk, E. (2013). Restoring macrophyte diversity in shallow temperate lakes vs. biotic versus abiotic constraints. *Hydrobiologia*, 710, 23–37.

Bakker, E.S., van Donk, E. & Immers, A.K. (2016). Lake restoration by in-lake iron addition: a synopsis of iron impact on aquatic organisms and shallow lake ecosystems. *Aquatic Ecology*, 50, 121–135.

Baastrop-Spohr, L., Kragh, T., Petersen, K., Moeslund, B., Schou, J.C. & Sand-Jensen, K. (2016). Remarkable richness of aquatic macrophytes in 3-year old re-established Lake Fil, Denmark. *Ecological Engineering*, 95, 375–383.

Barbour, M.T., Swietlik, W.F., Jackson, S.K., Courtemanch, D.L., Davies, S.P. & Yoder, C.O. (2000). Measuring the attainment of biological integrity in the USA: a critical element of ecological integrity. *Hydrobiologia*, 422/423, 453–464.

Battarbee, R.W., Anderson, N.J., Jeppesen, E. & Leavitt, P.R. (2005). Combining palaeolimnological and limnological approaches in assessing lake ecosystem response to nutrient reduction. *Freshwater Biology*, 50, 1772–1780.

Battarbee, R.W., Anderson, N.J., Bennion, H. & Simpson, G.L. (2012). Combining limnological and palaeolimnological data to disentangle the effects of nutrient pollution and climate change on lake ecosystems: problems and potential. *Freshwater Biology,* 27, 2091–2106.

Battarbee, R.W., Simpson, G.L., Shilland, E.M., Flower, R.J., Kreiser, A., Yang, H. & Clarke, G. (2014). Recovery of UK lakes from acidification: An assessment using combined palaeoecological and contemporary diatom assemblage data. *Ecological Indicators,* 37, 365–380.

Bennion, H. & Simpson, G.L. (2011). The use of diatom records to establish reference conditions for UK lakes subject to eutrophication. *Journal of Paleolimnology*, 45, 469–488.

Beschta, R.L. & Ripple, W.J. (2016). Riparian vegetation recovery in Yellowstone: The first two decades after wolf reintroduction. *Biological Conservation*, 198, 93–103.

Bradshaw, A.D. (1996). Underlying principles of restoration. *Canadian Journal of Fisheries & Aquatic Sciences*, 53, 3–9.

Brookes, A. & Shields, F.D. (1996). *River channel restoration: Guiding principles for sustainable projects*. Wiley, Chichester, UK.

Champkin, J., Copp, G.H., Sayer C.D., Clilverd, H.M., George, L., Vilizzi, L., Godard, M.J., & Clarke, J. (in press 2018). Responses of fishes and lampreys to the re-creation of meanders in a small English chalk stream. *River Research & Applications.*

Cris, R., Buckmaster, S., Bain, C. & Bonn, A. (eds.) (2011). *UK Peatland Restoration - Demonstrating Success.* IUCN UK National Committee Peatland Programme, Edinburgh.

CSIRO (2008). Water availability in the Murray–Darling Basin. Report from CSIRO to the Australian Government. CSIRO, Canberra. Available at http://www.csiro.au/files/files/po0n.pdf

Cunningham, J.M., Calhoun, A.J.K. & Glanz, W.E. (2007). Pond-breeding amphibian species richness and habitat selection in a beaver-modified landscape. *Journal of Wildlife Management*, 71, 2517–2526.

Davies, P.M. (2010). Climate change implications for river restoration in global biodiversity hotspots. *Restoration Ecology*, 18, 261–268.

Duarte, C., Conley, D., Carstensen, J. & Sánchez-Camacho, M. (2009). Return to *Neverland*: Shifting baselines affect eutrophication restoration targets. *Estuaries & Coasts*, 32, 29–36.

Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.-I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.-H., Soto, D., Stiassny, M.L.J. & Sullivan, C.A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews,* 81, 163–182.

England, J. & Gurnell, A.M. (2016). Incorporating catchment to reach scale processes into hydromorphology assessment in the UK. *Water & Environment Journal*, 30, 22–30.

European Union (1992). Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Official Journal of the European Communities*, L206, 7–50.

European Union (2000). Directive 2000/60/EC of the European Parliament and the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities*, L327, 1–72.

Feld C.K., Birk S., Bradley D.C., Hering D., Kail J., Marzin A., Melcher, A., Nemitz, D., Pedersen, M.L., Pletterbauer, F., Pond, D., Verdonschot, F.M. & Friberg, N. (2011). From natural to degraded rivers and back again: a test of restoration ecology theory and practice. *Advances in Ecological Research*, 44, 119–209.

Fergus, C.E., Lapierre, J.F., Oliver, S.K. Skaff, N.K. Cheruvelil, K.S. Webster, K., Scott, C. & Soranno, P. (2017). The freshwater landscape: lake, wetland and stream abundance and connectivity at macroscales. *Ecosphere*, 8 (8).

Findlay C.S, & Houlahan J. (1997). Anthropogenic correlates of species richness in south-eastern Ontario wetlands. *Conservation Biology*, 11, 1000–1009.

Friberg, N., Angelopoulos, N.V., Buijse, A.D., Cowx, I.G., Kail, J., Moe, T.F., Moir, H., O'Hare, M.T., Verdonschot, P.F.M. & Wolter C. (2016). Effective river eestoration in the 21st Century: From trial and error to novel evidence-based approaches. *Advances in Ecological Research*, 55, 535–611.

Johnson, R.K., Battarbee, R.W., Bennion, H., Hering, D., Soons, M.B. & Verhoeven, J.T.A. (2010). Climate change: defining reference conditions and restoring freshwater ecosystems. In Kernan, M., Battarbee, R.W. & Moss, M. (eds), *Climate Change Impacts on Freshwater Ecosystems.* Wiley-Blackwell, Chichester, UK

Gell, P., Mills, K. & Grundell, R (2013). A legacy of climate and catchment change: the real challenge for wetland management. *Hydrobiologia*, 708, 133–144.

Grabowski, R.C., Surian, N. & Gurnell, A.M. (2014). Characterizing geomorphological change to support sustainable river restoration and management. *WIREs Water*, 1, 483–512

Gurnell, A.M., Boitsidis, A.J., Thompson, K. & Clifford, N.J. (2006). Seed bank, seed dispersal and vegetation cover: Colonization along a newly-created river channel. *Journal of Vegetation Science*, 17, 665–674.

Gurnell, A.M., Rinaldi, M., Belletti, B., Bizzi, S., Blamauer, B., Braca, G., Buijse, T., Bussettini, M., Camenen, B., Comiti, F., Demarchi, L., García de Jalón, D., González del Tánago, M., Grabowski, R.C., Gunn, I.D.M., Habersack, H., Hendriks, D., Henshaw, A.J., Klösch, M., Lastoria, B., Latapie, A., Marcinkowski, P., Martínez-Fernández, V., Mosselman, E., Mountford, J.O., Nardi, L., Okruszko, T., O'Hare, M.T., Palma, M., Percopo, C., Surian, N., van de Bund, W., Weissteiner, C. & Ziliani, L. (2016). A multi-scale hierarchical framework for developing understanding of river behaviour to support river management. *Aquatic Sciences*, 78, 1–16.

Howard, L.C., Wood, P.J., Greenwood, M.T. & Rendell, H.M. (2009). Reconstructing riverine paleo-flow regimes using subfossil insects (Coleoptera and Trichoptera): the application of the LIFE methodology to paleochannel sediments, *Journal of Paleolimnology*, 42, 453–466,

Hobbs, R.J., Higgs, E. & Harris, J.A. (2009). Novel ecosystems: implications for conservation and restoration. *Trends in Ecology & Evolution,* 24, 599–605.

Hughes, P.D.M., Mauquoy, D., Barber, K.E. & Langdon, P.G. (2000). Mire-development pathways and palaeoclimatic records from a full Holocene peat archive at Walton Moss, Cumbria, England. *The Holocene*, 10, 465–479.

Jeppesen E., Søndergaard, M., Lauridsen, T.L. Davidson, T.A., Liu, Z., Mazzeo, N., Trochine, C., Özkan, K., Jensen, H.S., Trolle, D., Starling, F., Lazzaro, X., Johansson, L.S., Bjerring, R., Liboriussen, L., Larsen, S.E., Landkildehus, F. & Meerhoff, M. (2012). Biomanipulation as a restoration tool to combat eutrophication: recent advances and future challenges. *Advances in Ecological Research*, 47, 411–487.

Johnson, R.K., Battarbee, R.W., Bennion, H., Hering, D., Soons, M.B. & Verhoeven, J.T.A. (2010). Climate change: defining reference conditions and restoring freshwater ecosystems. In Kernan, M., Battarbee, R.W. & Moss, M. (eds), *Climate Change Impacts on Freshwater Ecosystems.* Wiley-Blackwell, Chichester, UK

Kail J., Arle, J. & Jähnig S.C. (2012). Limiting factors and thresholds for macroinvertebrate assemblages in European rivers: Empirical evidence from three datasets on water quality, catchment urbanization, and river restoration. *Ecological Indicators*, 18, 63–72.

Kail, J., Brabec, K., Poppe, M. & Januschke, K. (2015). The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: a meta-analysis. *Ecological Indicators*, 58, 311–321.

Lake, P.S. (2012). Flows, floods, floodplains and river restoration. *Ecological Management & Restoration*, 13, 210–211.

Law, A., Gaywood, M.J., Jones, K.C., Ramsay, P. & Willby, N.J. (2017). Using ecosystem engineers as tools in habitat restoration and rewilding: beaver and wetlands. *Science of the Total Environment,* 605–606, 1021–1030.

Lorimer, J., Sandom, C., Jepson, P., Doughty, C., Barua, M. & Kirby, K.J. (2015). Rewilding: science, practice, and politics. *Annual Review of Environment & Resources*, 40, 39–62.

Luvuno L.B., Kotze, D.C. & Kirkman, K.P. (2016). Long-term landscape changes in vegetation structure: fire management in the wetlands of KwaMbonambi, South Africa, *African Journal of Aquatic Science*, 41, 279–288.

Madgwick, G., Emson, D., Sayer, C.D., Willby, N.J., Rose, N., Jackson, M.J. & Kelly, A. (2011). Centennial-scale changes to the aquatic vegetation structure of a shallow eutrophic lake and implications for restoration. *Freshwater Biology*, 56, 2620–2636.

McBride, A., Diack, I., Droy, N., Hamill, B., Jones, P., Schutten, J., Skinner, A. & Street, M. (2011). *The Fen Management Handbook* 1st edn, Scottish Natural Heritage, Perth.

McCaffrey, M. & Eby, L. (2016). Beaver activity increases aquatic subsidies to terrestrial consumers. *Freshwater Biology*, 61, 818–532.

McCarthy. T.S., Tooth, S., Kotze, D.C., Collins, N.B., Wandrag, G. & Pike, T. (2010). The role of geomorphology in evaluating remediation options for floodplain wetlands: the case of Ramsar-listed Seekoeivlei, eastern South Africa. *Wetlands Ecology & Management*, 18, 119–134.

Middleton, B.A. (2013). Rediscovering traditional vegetation management in preserves: Trading experiences between cultures and continents *Biological Conservation*, 158, 271–279.

Moran, S., Perreault, M. & Smardon, R. (2016). Finding our way: A case study of urban waterway restoration and participatory process*. Landscape & Urban Planning*, October.

Nassauer J.I. (2004). Monitoring the success of metropolitan wetland restorations: cultural sustainability and ecological function. *Wetlands*, 24, 756–765.

Navarro, L.M. & Pereira, H.M. (2012). Rewilding abandoned landscapes in Europe. *Ecosystems*, 15, 900–912.

Nummi P. & Holopainen, S. (2014). Whole-community facilitation by beaver: ecosystem engineer increases waterbird diversity. *Aquatic Conservation: Marine & Freshwater Ecosystems,* 24, 623–633.

O'Connell, J.L., Johnson, L.A., Beas, B.J., Smith, L.M., McMurry, S.T. & Haukos, D.A. (2013). Predicting dispersal-limitation in plants: Optimizing planting decisions for isolated wetland restoration in agricultural landscapes. *Biological Conservation*, 159, 343–354.

Osenberg, C.W., Sarnelle, O. & Cooper, S.D. (1997). Effect size in ecological experiments: the application of biological models in meta-analysis. *American Naturalist*, 150, 798–812.

Palmer, M.A., Menninger, H.L. & Bernhardt, E. (2010). River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? *Freshwater Biology*, 55, 205–222.

Palmer, M.A., Hondula, K.L. & Koch B.J. (2014). Ecological restoration of streams and rivers: Shifting strategies and shifting goals. *Annual Review of Ecology, Evolution & Systematics*, 45, 247–69.

Papworth, S.K., Rist, J., Coad, L. & Milner-Gulland, E.J. (2009). Evidence for shifting baseline syndrome in conservation. *Conservation Letters*, 2, 93–100.

Perkin, J.S., Gido, B.K., Cooper, A.R., Turner, T.F., Osborne, M.J., Johnson, E.R. & Mayes, K.B. (2015). Fragmentation and de-watering transform Great Plains stream fish communities. *Ecological Monographs*, 85, 73–92.

Perry, L.G., Reynolds, L.V., Beechie, T.J., Collins, M.J. & Shafroth, P.B. (2015). Incorporating climate change projections into riparian restoration planning and design. *Ecohydrology*, 8, 863–879.

Phillips, G., Bennion, H., Perrow, M., Sayer, C.D., Spears, B. & Willby, N. (2015). *A review of lake restoration practices and their performance in the Broads National Park, 1980–2013.* Report for Broads Authority & Natural England, Norwich.

Phillips, G., Willby, N. & Moss, B. (2016). Submerged macrophyte decline in shallow lakes: What have we learnt in the last forty years? *Aquatic Botany*, 135, 37–45.

Pittock J. & Finlayson C.M. (2011). Australia's Murray Darling Basin: freshwater ecosystem conservation options in an era of climate change. *Marine & Freshwater Research*, 62, 232–243.

Puttock, A., Graham, H.A., Cunliffe, A.M., Elliott, M. & Brazier, R.E. (2017). Eurasian beaver activity increases water storage, attenuates flow and mitigates diffuse pollution from intensively-managed grasslands. *Science of the Total Environment*, 576, 430–443.

Ramstead, K.M., Allen, J.A. & Springer, A.E. (2012). Have wet meadow restoration projects in the Southwestern U.S. been effective in restoring geomorphology, hydrology, soils, and plant species composition? *Environmental Evidence*, 1, 11.

Rinaldi, M., Gurnell, A.M., González del Tánago, M., Bussettini, M. & Hendriks, D. (2016). Classification of river morphology and hydrology to support management and restoration. *Aquatic Sciences*, 78, 17–33.

Sayer C.D., Davidson, T.A., Jones, I.J., Langdon, P.G. (2010). Combining contemporary ecology and palaeolimnology to understand shallow lake ecosystem change. *Freshwater Biology*, 55, 487–499.

Sayer, C.D., Andrews, K., Shilland, E., Edmonds, N., Edmonds-Brown, R., Patmore, I., Emson, D. & Axmacher, J.A. (2012). The role of pond management for biodiversity conservation in an agricultural landscape. *Aquatic Conservation: Marine & Freshwater Ecosystems*, 22, 626–638.

Sayer, C.D. (2014). Conservation of aquatic landscapes: ponds, lakes, and rivers as integrated systems. *WIRE's: Water*, 1, 573–585.

Schumm, S.A. (1994). Erroneous perceptions of fluvial hazards. *Geomorphology*, 10, 129–138.

Stendera, S. & Johnson, R. K. (2008). Tracking recovery trends of boreal lakes: use of multiple indicators and habitats. *Journal of the North American Benthological Society*, 27, 529–540.

Stroh, P.A., Hughes, F.M.R., Sparks, T.H., Mountford, O. & Mountford, J.O. (2012). The influence of time on the soil seed bank and vegetation across a landscape-scale wetland restoration project. *Restoration Ecology*, 20, 103–112.

Tansley, A.G. (1939). *The British Islands and Their Vegetation*. Cambridge University Press, Cambridge.

Trexler, J.C. (1995). Restoration of the Kissimmee River: A conceptual model of past and present fish communities and its consequences for evaluating restoration success. *Restoration Ecology*, 3, 195–210.

Thompson, M.S.A., Brooks, S.J., Sayer, C.D., Woodward, G., Axmacher, J.C., Perkins, D. & Gray, C. (in press 2018). Large woody debris 'rewilding' restores biodiversity in riverine food webs. *Journal of Applied Ecology.*

Verdonschot, P.F M., Spears, B.M., Feld, C.K., Brucet, S., Keizer-Vlek, H., Borja, A., Elliott, M., Kernan, M. & Johnson, R.K. (2013). A comparative review of recovery processes in rivers, lakes, estuarine and coastal waters. *Hydrobiologia*, 704, 453–474.

Vermaat, J., Wagtendonk, A.J., Brouwer, R., Sheremet, O., Ansink, E., Brockhoff, T., Plug, M., Hellsten, S., Aroviita, J., Tylec, L., Gielczewski, M., Kohut, L., Brabec, K., Haverkamp, J., Poppe, M., Boeck, K., Coerssen, M., Segersten, J. & Hering, D. (2016). Assessing the societal benefits of river restoration using the ecosystem services approach. *Hydrobiologia*, 769, 121–135.

Wals A.E.J. (2007). Learning in a Changing World and Changing in a Learning World: Reflexively Fumbling Towards Sustainability. *South African Journal of Environmental Education*, 24, 35–45.

Weinhold, C.E. & van der Valk, A.G. (1989). The impact of drainage on the seed banks of northern prairie wetlands. *Canadian Journal of Botany*, 67, 1878–1884.

Weisstenier, C.J., Bouraoui, F. & Aloe, A. (2013). Reduction of nitrogen and phosphorus loads to European rivers by riparian buffer zones. *Knowledge & Management of Aquatic Ecosystems*, 408, 08.

Whitehead, P.G., Wilby, R.L., Butterfield, D. & Wade A.J (2006). Impacts of climate change on in-stream nitrogen in a lowland chalk stream: An appraisal of adaptation strategies. *Science of the Total Environment*, 365, 260–273.

Whitehouse, N., Langdon, P., Bustin, R. & Galsworthy, S. (2008). Fossil insects and ecosystem dynamics in wetlands: implications for conservation and management. *Biodiversity & Conservation*, 17, 2055–2078.

Willby N.J. (2011). From metrics to Monet: The need for an ecologically meaningful guiding image (Editorial). *Aquatic Conservation: Marine & Freshwater Ecosystems*, 21, 601–603.

Wright J.F., Sutcliffe, D.W & Furse, M.T. (eds) (2000) *Assessing the biological quality of fresh waters. RIVPACS and other techniques. Proceedings of an International Workshop, Oxford, September 1998.* Freshwater Biological Association, Ambleside, UK.

Figures for OUP Freshwater Restoration chapter – Sayer et al.

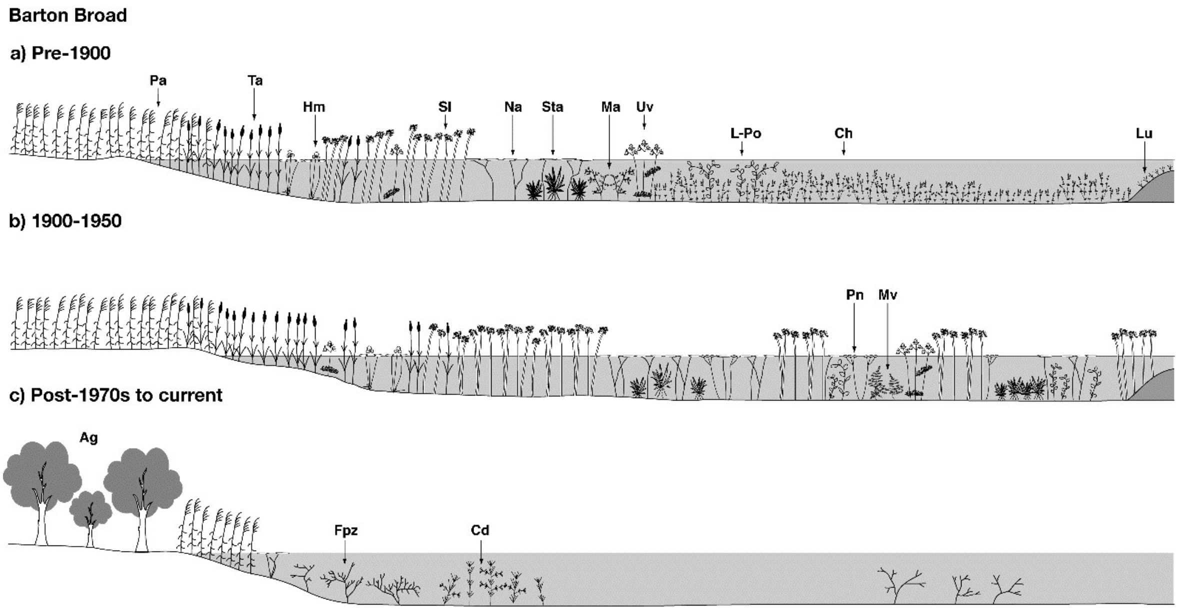

Figure 1.1. Reconstruction of macrophyte spatial relationships in Barton Broad over three time periods; pre-1900 (a), 1900-1950 (b), 1970s-current (c). Codes to plant names: Pa – *Phragmites australis*, Ta – *Typha angustifolia*, Hm – *Hydrocharis morsus-ranae*, Sl – *Schoenoplectus lacustris*, Na – *Nymphaea alba*, Sta – *Stratiotes aloides*, Ma – *Myriophyllum alterniflorum*, Uv – *Utricularia vulgaris*, L-Po – Large, broad-leaved *Potamogeton* taxa (e.g. *P. lucens, P. praelongus, P. alpinus*), Ch – *Chara* spp., Lu – *Littorella uniflora*, Pn – *Potamogeton natans*, Mv – *Myriophyllum verticillatum*, Ag – *Alnus glutinosa*, Fpz – Fine-leaved *Potamogeton* taxa (e.g. *P. pectinatus, P. pusillus*) and *Zannichellia palustris*, Cd – *Ceratophyllum demersum* (from Madgwick et al 2011).

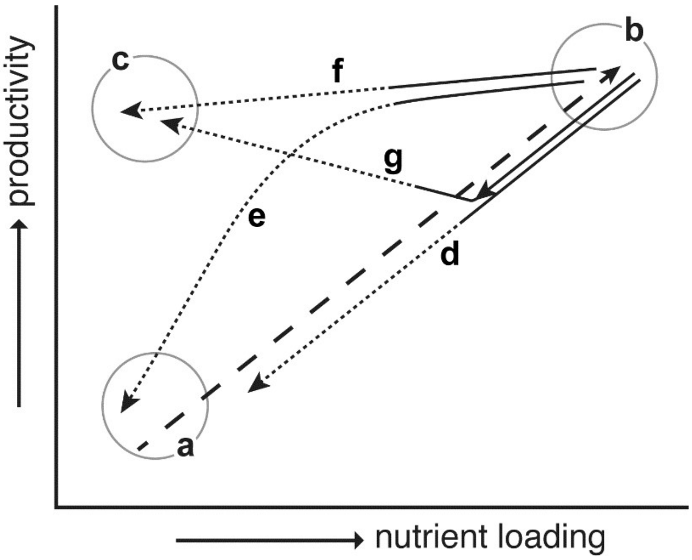

Figure 1.2. Conceptual diagram using a combination of palaeolimnological data (dashed line), contemporary long-term monitoring data (solid lines) and future conjecture (dotted lines) to show idealised changes in the past, present and future relationship between nutrient loading and productivity for European lakes recovering from eutrophication. Point (a) indicates the reference state and the target endpoint following restoration; point (b) indicates the point of intervention to reduce nutrient loading; and (c) indicates a more probable potential endpoint in cases where recovery to the past reference (a) is prevented by the enriching effects of climate change. Arrow d) represents a simple trajectory back towards the reference state, e) represents delayed recovery towards the reference state (e.g. due to internal P loading), and f) and g) represent deflected trajectories away from the reference and towards a new endpoint (from Battarbee et al 2012).

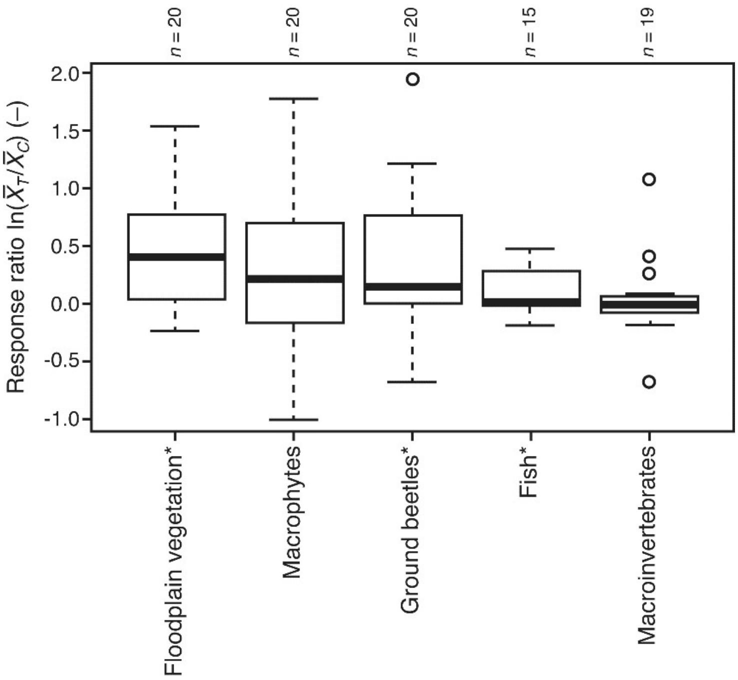

Figure 1.3. Effects of restoration on species richness (5 organism groups) for European rivers as reflected by the response ratio of Osenberg et al (2017) which relates the value of a restored section (X_T) to a degraded control section (X_C). Mean values that are significantly different to zero (*t*-test, p<0.05) are marked with an asterisk and positive values are indicative of positive restoration responses (from Friberg et al 2016).

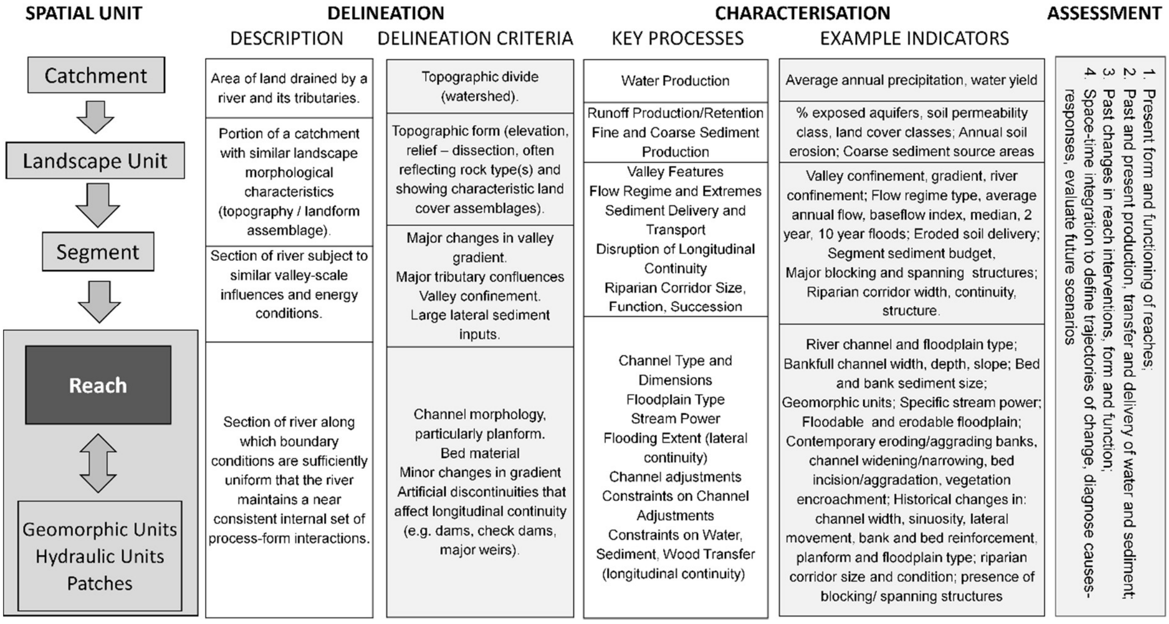

Figure 1.4. The multi-scale REFORM framework, that follows delineation, characterisation and assessment-diagnosis phases in order to develop understanding of how the character of individual river reaches adjust to processes and pressures operating at catchment to patch scales and change from the past through the present to the future (based on concepts explained in Gurnell et al. 2016).

Figure 1.5. Shooting Close Pond, a small farmland pond in eastern England, UK, before (a), during (b) and two years after restoration (c) by scrub and sediment removal in September 2014.

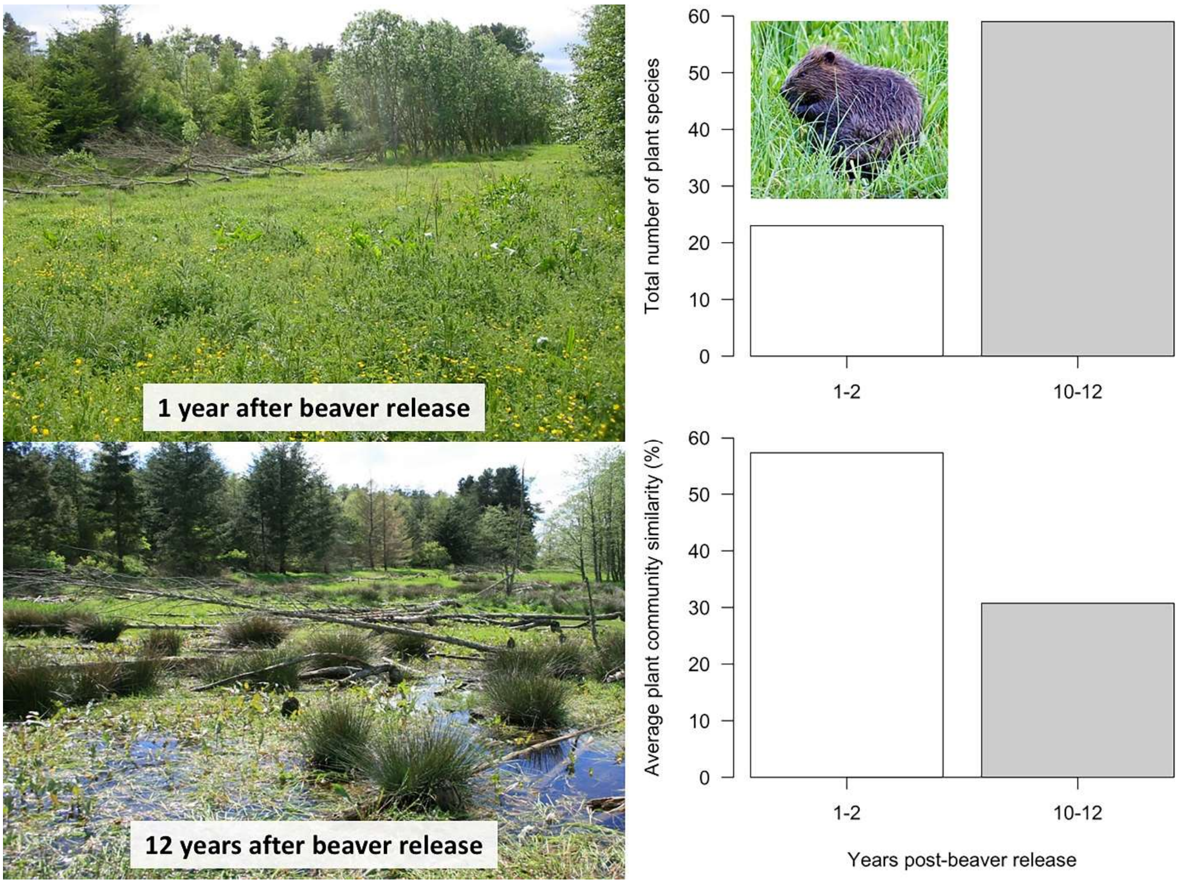

Figure 1.6. Overview of results from a long-term European beaver (*Castor fiber*) release study in Blairgowrie, eastern Scotland showing the study site 1 year and 12 years post release (left) and accompanying changes to wetland vegetation (right). From Law et al (2017).