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SEVENTH FRAMEWORK PROGRAMME

WISER

Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery

DELIVERABLE

Deliverable D6.4-3: Final report on impact of catchment scale processes and climate change on cause-effect and recovery-chains

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CO	Confidential, only for members of the consortium (including the Commission Services)	

Content

Summary

Introduction

Catchment wide integrated basin management requires knowledge on cause-effect and recovery chains within water bodies as well as on the interactions between water bodies and categories. In the WISER WP6.4 recovery processes in rivers, lakes and estuarine and coastal waters were evaluated. The major objectives were:

- To analyse and compare (cause-effect and) recovery chains within water categories based on processes and structural and functional features.
- To detect commonalities among different chains in different water categories. Thus, to compare recovery chains between water categories.
- To link recovery chains to over-arching biological processes and global change.
- To develop a method to combine recovery effects in a summarising 'catchment' metric.

The main stressors studied to reach these objectives were acidification, eutrophication and hydromorphological changes.

Methods

To compare recovery-chains within water bodies and between water categories information was extracted from published reports and peer-reviewed papers. Apart from a variety of about 20 major reviews, three major sources of information were included. For rivers 370 papers were reviewed and 168 papers were analysed by Feld et al. (2011). For lakes 302 lake-equivalent recovery case studies for which eutrophication was the major stressor were analysed in detail by Spears et al. (2011). Also, 30 peer-reviewed publications reporting on the management of 41 eutrophic lakes were reviewed in more detail. For estuarine and coastal waters the review of 51 studies by Borja et al. (2010) was the major information source.

Results

Degradation

Rivers integrate the adverse effects of various activities on land and are, therefore, often simultaneously affected by multiple stressors arising from agriculture, deforestation, urbanization, storm water treatment, flow regulation and water abstraction (Palmer et al. 2010). Globally, lake ecosystems are mainly being affected by eutrophication (intensive agricultural land use) and physical habitat modification of their shoreline, while estuaries and wetlands constitute the ultimate sink for nutrients and other sources of pollution and contaminants originating from entire river basins. Furthermore, many estuarine and coastal waters are being physically modified, for instance, for flood protection purposes and navigation. The conceptual models (DPSIRR-chains) of the different water categories are hard to compare. Striking is the difference in the level of detail between rivers and lakes (high) on the one hand and the marine

ecosystems (low) on the other. This difference probably has to do with the scale of degradation in rivers and lakes, where it is easier to find/deduct pathways of ecosystem response.

Recovery Concepts

The Driver-Pressure-State-Impact-Response-Recovery (DPSIRR) scheme provides a framework to link socio-economy with ecology. Literature was searched for existing DPSIRR-chains for the three water categories. Such conceptual models on the recovery of river, lake and estuarine and coastal ecosystems were scarce and fragmented. Such models lacked for the marine systems were quite one-sided, focusing on eutrophication, for lakes and quite specific for certain measures in rivers. Comparison and integration of DPSIRR-chains is up date impossible.

Restoration Measures

In rivers most measures target the morphology of the stream stretch or the instream habitats. Few only are related to reduction of nutrient input. On the contrary, in lakes the most common measure targeted is the reduction of nutrient levels, especially phosphate. For acidification of streams and lakes, liming is commonly used in some countries (e.g. Sweden) for mitigating the effects of acidification, while decreased emission and deposition of acidifying compounds is a more cost-effective, long-term measure of remediation. Measures are not often taken directly in estuarine and coastal waters, these much more relate to measures taken inland through legislation on nutrient reduction. These observations supported our initial hypothesis that “at a catchment scale, nutrient stress affecting functional (production/decomposition) processes will be more important in lakes and marine systems, while hydromorphological stress affecting habitat availability will be more important in rivers”.

Recovery: Data availability and processing

In rivers and lakes a substantial amount of monitoring data are available. In estuarine and coastal waters such data are scarce. Despite the number of monitored recovery cases, each one seems to stand alone, as monitoring schemes were set-up for local situations and to answer partial questions. By contrast, for acidification liming efforts often target individual lakes or streams, but even large-scale liming of catchments has been performed. Furthermore, in many, many cases data on recovery just lack and this is quite alarming! Not only is the amount of available data surprisingly low, the composition of the available data is often very limited and does not allow the evaluation and generalisations of improvements and eventually of successes. The huge investments in recovery of surface waters require control of the ecological effects. Therefore, restoration monitoring should become mandatory. Only by frequent monitoring of biological and abiotic changes after restoration will restoration practitioners and scientist be able to evaluate the success of the restoration measure and eventually of the investment done.

Recovery: Organism groups

The majority of restoration studies in rivers and in estuarine and coastal ecosystems have focused on macroinvertebrates. In rivers also fish are important indicators. In lakes phytoplankton is the BQE studied most extensively. The difference in indicator groups used goes back to the causes of degradation. In lakes eutrophication is most important and phytoplankton best reflects the nutrient status of the lake over time. In rivers most degradation goes with hydromorphological change. Macroinvertebrates and fish respond strongly to these types of changes. The choice of macroinvertebrates as indicators of degradation in estuarine and coastal waters is less obvious as eutrophication and organic load are most common causes of degradation along with bottom disturbances. The latter would best be reflected in macroinvertebrate responses, while for tracking responses due to elevated nutrients a primary producer like phytoplankton is probably the most sensitive. The confounding factor in estuarine and coastal waters for phytoplankton is water movement, i.e. water movement reduces the indicative value of phytoplankton.

Recovery: Time-scale

Although, analyses in the different reviews do not address full recovery', authors do give indications on 'full recovery' based on estimates. Marine ecosystems may take between 35 and 50 years to recover. Recovery after weir removal may take as long as 80 years. Recovery after riparian buffer installment may take at least 30-40 years. Recovery after liming can be rapid (< 1 year) for some response indicators like water chemistry and organisms with resting stages and/or with high dispersal (phytoplankton, zooplankton), whilst recovery of other groups such as fish can take much longer (> 10 years), Despite the fact that they do not indicate 'full recovery' we compared recovery times between the three water categories as mentioned in the different reviews. In marine ecosystems benthic invertebrates and macrophytes have the potential to recover within months (in two studies on recovery of sediment disposal) and fish within one year. When only marine studies that recover from eutrophication are included, recovery times for macroinvertebrates varied between >3 years and >6 years. Although in some cases recovery can take <5 years, especially for the short-lived and high-turnover biological components, full recovery of estuarine and coastal ecosystems from over a century of degradation can take a minimum of 15–25 years for attainment of the original biotic composition, diversity and complete functioning may lag far beyond that period. In lakes recovery time from eutrophication for macroinvertebrates varied between 10 and 20 years. As in marine ecosystems recovery of macrophytes (2 to >40 years) and fish in lakes (2 to >10 years) be relatively fast. Response times for organism groups in rivers are lacking, because the literature rarely includes post hoc monitoring of more than 5 years. Also, the fact if biological response in rivers occurs within short term is undecided. The potential benefits of most in-stream structures will be short-lived (<10 years) unless coupled with riparian planting or other process-based restoration activities supporting long-term recovery of key ecological and physical processes.

In both rivers and lakes the success rate of restoration measures appears to be much higher for the abiotic conditions than for the biotic indicators. Since eutrophication is considered to be the most important pressure in rivers and lakes, only this is not addressed in rivers, this might be a major cause. Especially, the response of macroinvertebrates in rivers is questionable; some studies mention recovery times, while others question recovery of macroinvertebrates completely. In lakes internal nutrient loading often delays recovery. By contrast, responses to liming are often rapid, with some indicators responding almost immediately. For example, following the “shock” effect of liming on phytoplankton biomass, production and species richness, within only a few months new phytoplankton species were observed (Svensson et al. 1995). For lakes recovering naturally from acidification, responses are much slower and results often equivocal. For example, Stendera and Johnson (2008) showed that responses indicator-specific, with chemical changes occurring very rapidly, followed by phytoplankton, and littoral benthic invertebrates. Sublittoral and profundal benthic invertebrate assemblages did not show a response to decreased deposition of acidifying compounds.

Recovery: Failure or delay in response

Several major reasons return in many publications on recovery failure or delay:

- Spatial scale must be large enough (catchment).
- Temporal scale: there is time needed for recovery.
- Multistressors present: mostly only one or a few stressors were tackled, others forgotten.
- Confounding abiotic processes affect recovery, such as upstream ‘hidden’ stressors, internal P loading, and biological interactions, like the early arrival of non-native species, but also climate change effects, effects of management and maintenance.
- Distance from source populations and lack of connectivity results in dispersal limitations and colonisation barriers.
- There is no guiding monitoring that makes evaluation along the development and redirection of measures possible.

Recovery: Shifting baselines

It is difficult to judge whether the concept of shifting baselines is part of the reality of ecosystems developments as proof is hard to find. Even in the coastal and estuarine examples it is questionable whether the responses are due to alternative states or due to overlooked other stressors. Often in many lake examples the latter is the case. For lakes the sediment record provides a valuable tool for establishing reference conditions. Sub-fossil assemblages of organisms such as diatoms which are sensitive to environmental stress can be used to determine both the degradation and the recovery process. In terms of lake management, while the baseline reamains an important concept, it should also be recognised that the recovery trajectory may not simply represent the reverse process of the degradation pathway and that the reference state may perhaps never be achievable in some lakes.

Recovery: Effects of biological interactions

Restoring the appropriate habitat (both structural and chemical) is still the main component of aquatic ecosystem restoration efforts. Although the importance of establishing the suitable abiotic conditions is stressed by a multitude of studies, the awareness that other factors should be considered as well is apparent in recent recommendations on freshwater restoration. There are several, more or less connected issues that are repeatedly stressed in a multitude of studies:

- Incorporating the spatial and temporal scale (i.e. maximum and minimum) of the habitat and the connectivity between the various habitat patches, including both abiotic and biotic components;
- Incorporating the knowledge of source populations and dispersal ability or constraints in predicting restoration outcome. However, few studies attempt to match this ecological background with empirical data.
- Incorporating mitigating measures to prevent non-native species to colonise and set priority effects.

Recovery: Impacts of climate and global change

A range of biological management practices (especially fishery management) and extreme weather events were identified as key factors that were responsible for slowing down or contradicting recovery processes. Alterations in nutrient concentrations and biogeochemical cycling at the sediment-water interface, following nutrient management, can influence the magnitude and timing of nutrient delivery to downstream ecosystems. This phenomenon is likely to be highly sensitive to changes in local weather conditions associated with climate change.

Research gaps

In summary, there is need for the following research efforts;

- Need for empirical understanding of ecological responses.
- Need for more comprehensive and long-term monitoring to underpin quantitative assessment of management measures. Indeed, when robust BACI studies are not available, most studies must rely on correlative data. Here having access to data from restored sites, unperturbed reference sites (the target of restoration) and unrestored perturbed sites (the lower anchor to include effects such as climate-induced change) are generally needed to best assess the efficacy of restoration effort.
- Need to quantitatively assess cause-effect relationships during the recovery process.
- Need to test if indicators/metrics calibrated to detect degradation are sensitive to detect improvement.
- Need for case studies relevant to WFD targets.
- Need for specific knowledge on certain BQEs in certain water categories.
- Need for knowledge on maintenance, and recurring management.
- Need for knowledge on the most important factor(s) for recovery and their interactions.
- Need for knowledge on shifting baselines and thresholds.

Conclusions

Restoration ecology is just evolving. The huge amount of literature evaluated brings up one major conclusion. Restoration is a site, time and organism group specific activity. Generalisations on recovery processes are up to date hard to make. Despite the multitude of studies that provided theoretical frameworks, guidelines, research needs and issues that are important for freshwater restoration, only few studies provide evidence of how this ecological knowledge might enhance restoration success.

Goals of restoration projects typically encompass a multitude of objectives (species groups, ecological, cultural and landscape values) and a multitude of measures. Thus, evaluation of the response of a single factor to a single measure tends to be difficult.

Another major bottleneck is the lack of sufficient monitoring allowing for insufficient learning from both successful and unsuccessful restorations. However, the frequently occurring general recommendation in proposed guidelines for restoration projects, including appropriate monitoring and publishing of the results, could help to gain insight into the processes important to successful restoration.

Another problem is related to the many detected effects that occur only in the short-term and at the local (site) scale, which raises the question of appropriate scaling for restoration. Knowledge of which spatial or temporal scales are relevant is for the most part lacking, but several review studies supported the hypothesis that the local scale is inappropriate to achieve long-term measurable improvements.

Chapter 0. Introduction

Context

Module 6, "***Integration and Optimisation***", addresses the question how different Biological Quality Elements (BQEs) and water categories respond to degradation and to rehabilitation or restoration. Ecological assessments frequently use several BQEs collected from a number of systems (lakes, streams, coastal areas) and habitats within these systems (e.g. pelagic and benthic, littoral and profundal or riffles and pools). Two innovative aspects of the WFD are:

- the recognition that aquatic systems are not isolated entities, but are nested within a catchment,
- that the use of multiple BQEs may increase our understanding of the effects that humans are having on aquatic resources.

The use of complementary indicator variables can strengthen inference-based assessments (see WP6.1) of human-generated effects as well as help to distinguish the effects of multiple stressors. This latter conjecture implies that different BQEs may respond similarly to some stressors (e.g. high redundancy used to strengthen the inference model) but differently to other stressors (low redundancy, but indicating the presence of other stressors). To design and implement cost-effective management programmes and to scientifically underpin conceptual models of aquatic ecosystem responses to human-induced changes more knowledge of the response signatures of BQEs is required (see WP6.3), as well as an understanding of the uncertainties (see WP6.1) associated with the use of different BQEs. Moreover, in designing robust monitoring programmes consideration should be given not only to the selection of BQEs (see WP6.2) to test specific hypotheses of change (i.e. stress-response relationships) but also to the system and the habitat that may provide the most robust measure (high precision/low error) of change.

More in detail Work Package 6.4 "***Comparison of recovery processes between water categories***" will focus on comparing the cause-effect-recovery chains for lakes, rivers and marine ecosystems, taking into account processes and functional features in different ecosystems and over-arching biological processes of connectivity and metapopulation dynamics. Special focus will be placed on the use of species traits and functional information in cross-water category comparisons.

Objectives

Catchment wide integrated basin management requires knowledge on cause-effect and recovery chains within water bodies as well as on the interactions between water bodies and categories.

Approaches include processes related to biology (connectivity, metapopulation and dispersal) and global change (climate change, land and water use). The main stressors studied in this WP were acidification, eutrophication and hydromorphological changes. The WP 6.4 objectives were:

- To analyse and compare (cause-effect and) recovery chains within water categories based on processes and structural and functional features.
- To detect commonalities among different chains in different water categories. Thus, to compare recovery chains between water categories.
- To link recovery chains to over-arching biological processes and global change.
- To develop a method to combine recovery effects in a summarising ‘catchment’ metric.

The concept: DPSIRR-chain

The Driver-Pressure-State-Impact-Response-Recovery (DPSIRR) scheme provides a framework to link socio-economy with ecology (EEA 2007, Feld et al. 2011). The framework can best be illustrated by an example of reasoning: population growth resulting in an increasing food demand is a Driver of agricultural land use. The intensive application of fertilisers in agricultural crops is linked with eutrophication and morphologically modified and hydrologically regulated water flows (Pressure). The first has a stimulating direct effect on the growth of macrophytes and algae and also indirectly and negatively affects the aquatic fauna (fish, benthic invertebrates) as soon as decomposers start depleting oxygen and causing water quality deterioration (State). As a consequence, the aquatic ecosystem is being disrupted, sensitive taxa disappear and a few tolerant taxa become dominant in the system (Impact). To reverse degradation and to improve the ecological status, restoration and mitigation measures are required. Best-practice agriculture (Response), for instance, might reduce the amount of fertilisers and hydromorphological conditions might be actively restored (Response) to a more diverse habitat and flow regime. These measures in turn should have a positive Impact on the biota (Recovery).

Method

In WP6.4 three major tasks were defined:

1. Comparison of cause-effect and recovery-chains within water bodies
2. Comparison of cause-effect and recovery-chains between water categories
3. Linking catchment scale processes and global change to cause-effect and recovery-chains

At first, the ideas at the start of WISER were that this WP would build on the results of Module 5 and also incorporate knowledge from analyses of data collected in Modules 3 and 4 and complete water body assessment in WP6.2. In the end, this was especially the case for the link between WP5.1 and WP6.4 for Rivers, for Module 4 and WP6.4 for Marine waters and to a limited extend for Module 3 and WP6.4 for Lakes.

To compare cause-effect and recovery-chains within water bodies and between water categories (Tasks 1 and 2) WP6.4 started to collect metadata for lakes, rivers and transitional/coastal waters for the BQEs diatoms, macroinvertebrates, fishes and macrophytes from literature and existing databases. The original idea was to use metadata analysis, functional traits, metrics, and multivariate and statistical techniques to make cause-effect and recovery chains for different drivers of change comparable between water categories. The analysis would follow the hypotheses: At a catchment scale, we anticipate that nutrient stress affecting functional (production/decomposition) processes will be more important in lakes and marine systems, while hydromorphological stress affecting habitat availability will be more important in rivers.

After several attempts and meta data analysis and comparisons it became clear that the data were far too incomplete and incomparable to perform the task based on existing data. The original idea was to compare the parameters selected in the previous step for cause-effect and recovery-chains within water bodies and between different water categories.

The new approach chosen for rivers, lakes and coastal waters was to extract the information needed from published reports and peer-reviewed papers. With this adapted approach we tried to keep the original line of reasoning but selected the parameters from literature sources.

The key words or items in this literature study were:

Water category: river, lake, transitional/coastal water, estuary/estuarine

Stressor: acidification, eutrophication, hydromorphological change (water level change, water flow, bank profile, bottom substrate), shoreline modification, nutrients

Recovery process: recovery chain, habitat availability, restoration, rehabilitation, trajectory improvement, biology, nature of interaction between multiple stressors (antagonistic, neutral, additive or synergistic), long-term effects, reduced S/N deposition

Organism group: macroinvertebrate, fish, macrophyte, algae/diatoms

Global change: climate change, land use, water use

Biology: connectivity, interaction, dispersal, metapopulation, functioning, production, decomposition, biological processes,

Community characteristic: functional information, functional/trait indices, metrics, diversity/evenness indices, community indices, assessment techniques, structural community, assemblage

Methodology: before/after, control/impact, space-for-time substitution, time series, palaeolimnology

Strength of relation: characterize each relation between restoration method / recovery time and biotic response by statistical parameters (r, r^2 or whatever) to enable a meta-analysis

The first results on the comparison of cause-effect and recovery-chains within water bodies (Task 1) were incorporated in the second Deliverable 6.4.2 "*Report on the differences between cause-effect-recovery chains of different drivers*".

For Task 3 a separate study was performed that defined the relevant biological processes of connectivity and metapopulation dynamics. The results of this study were described in Deliverable 6.4.1 "*Literature report on biological processes on catchment scale, such as connectivity, dispersal and metapopulation dynamics*". The effect of these overarching processes of biological interactions and also of global change on cause-effect and recovery-chains will be incorporated in this Deliverable.

The overall synthesis of WP6.4 was based on the foregoing literature inventories and studies. The current third Deliverable 6.4.3 provides the "Final report on impact of catchment scale processes and global change on cause-effect and recovery-chains".

To compare recovery-chains within water bodies and between water categories information was extracted from published reports and peer-reviewed papers. Apart from a variety of about 20 major reviews, three major sources of information were included. For rivers 370 papers were reviewed and 168 papers were analysed by Feld et al. (2011). For lakes 302 lake-equivalent recovery case studies for which eutrophication was the major stressor were analysed in detail by Spears et al. (2011). Also, 30 peer-reviewed publications reporting on the management of 41 eutrophic lakes were reviewed in more detail. Three lakes were included more than once in the literature (maximum of 3 occurrences) resulting in 46 lake equivalent case studies (LECs). The publications were not selected randomly. Instead effort was taken to ensure that at least 3 publications were reported for a range of eutrophication management measures. These pre-defined management measures, along with the number and percentage of LECs for which data on each measure was reported in the meta-dataset, are reported (Figure 1). The LECs reported data from 9 countries (Figure 2) and were dominated by very shallow (56 % LECs < 3 m mean depth) and shallow (41.3 % of LECs 3-15 m mean depth) WFD lake types. For acidification, 30 published manuscripts were reviewed focusing on biological recovery at previously acidified lakes and streams (21 papers on lakes only, 7 on streams only and 2 covering both). Most papers assessed recovery following reductions in atmospheric deposition of acidifying compounds but several reported on the outcome of liming manipulations. A total of 419 lakes and141 streams were included in the 30 publications although numerous papers featured the same sites. For estuarine and coastal waters the review of 51 studies by Borja et al. (2010) was the major information source.

Chapter 1. Degradation

Driver-Pressure-State-Impact-Response chains

Growth of the human population inevitably results in greater use of natural resources and emission of pollutants, which in turn puts more pressure on the environment. Furthermore, the magnitude and spatial reach of human alterations of the land surface are continuously increasing (Turner et al. 1990, Lambin et al. 1999). However, the control of drivers (e.g. population growth) is commonly not the focus of specific water management programmes due to socioeconomic interests at the national and catchment scales.

More attention is given to primary and secondary pressures, as illustrated by some marine, lake and river examples.

Rivers

Rivers are strongly influenced by their surroundings at multiple scales (Allan et al. 1997, Fausch et al. 2002, Schlosser 1991, Townsend et al. 2003). River ecologists have long recognized that rivers and streams are influenced by the catchment(s) through which they flow (Hynes 1975, Vannote et al. 1980). Human activities at the catchment scale are a principal threat to the ecological integrity of river ecosystems, impacting habitat, water quality, and the biota via numerous and complex pathways (Allan et al. 1997, Strayer et al. 2003, Townsend et al. 2003). In addition to its direct influences, land use interacts with other anthropogenic drivers that affect the integrity of river and stream ecosystems, including climate change (Meyer et al. 1999), invasive species (Scott & Helfman 2001), and dams (Nilsson & Berggren 2000). Furthermore, there is a widespread recognition of the extent and significance of changes in land use and land cover worldwide (Meyer & Turner 1994) on the river state (Table 1).

Rivers are also affected by airborne pollutants affecting whole regions, exemplified by the deleterious effects of acidifying compounds (N and S) emitted in the burning of fossil fuels on biodiversity. In particular, waters running through poorly buffered catchments are most strongly affected; the most serious effects have been found in coniferous regions with lime-deficient soils. Moreover, since catchments with poorly buffered soils are not productive for agriculture, acidification is often the most important (single) affecting the integrity of streams.

Table 1. Principal mechanisms by which land use influences stream ecosystems (Allan 2004).

Environmental factor	**Effects**	**References**
Sedimentation	Increases turbidity, scouring and abrasion; impairs	Burkhead & Jelks 2001,

	substrate suitability for periphyton and biofilm production; decreases primary production and food quality causing bottom-up effects through food webs; in-filling of interstitial habitat harms crevice-occupying invertebrates and, gravel-spawning fishes; coats gills and respiratory surfaces; reduces stream depth heterogeneity, leading to decrease in pool species	Hancock 2002, Henley et al. 2000, Quinn 2000, Sutherland et al. 2002, Walser & Bart 1999, Wood & Armitage 1997
Nutrient enrichment	Increases autotrophic biomass and production, resulting in changes to assemblage composition, including proliferation of filamentous algae, particularly if light also increases; accelerates litter breakdown rates and may cause decrease in dissolved oxygen and shift from sensitive species to more tolerant, often non-native species	Carpenter et al. 1998, Delong & Brusven 1998, Lenat & Crawford 1994, Mainstone & Parr 2002, Niyogi et al. 2003
Contaminant pollution	Increases heavy metals, synthetics, and toxic organics in suspension associated with sediments and in tissues; increases deformities; increases mortality rates and impacts to abundance, drift, and emergence in invertebrates; depresses growth, reproduction, condition, and survival among fishes; disrupts endocrine system; physical avoidance.	Clements et al. 2000, Cooper 1993, Kolpin et al. 2002, Liess & Schulz 1999, Rolland 2000, Schulz & Liess 1999, Woodward et al. 1997
Hydrologic alteration	Alters runoff-evapotranspiration balance, causing increases in flood magnitude and frequency, and often lowers base flow; contributes to altered channel dynamics, including increased erosion from channel and surroundings and less-frequent overbank flooding; runoff more efficiently transports nutrients, sediments, and contaminants, thus further degrading in-stream habitat. Strong effects from impervious surfaces and stormwater conveyance in urban catchments and from drainage systems and soil compaction in agricultural catchments.	Allan et al. 1997, Paul & Meyer 2001, Poff & Allan 1995, Walsh et al. 2001, Wang et al. 2001
Riparian clearing/canopy opening	Reduces shading, causing increases in stream temperatures, light penetration, and plant growth; decreases bank stability, inputs of litter and wood, and retention of nutrients and contaminants; reduces sediment trapping and increases bank and channel erosion; alters quantity and character of dissolved organic carbon reaching streams; lowers retention of benthic organic matter owing to loss of direct input and retention structures; alters trophic structure	Bourque & Pomeroy 2001, Findlay et al. 2001, Gregory et al. 1991, Gurnell et al. 1995, Lowrance et al. 1984, Martin et al. 1999, Osborne & Kovacic 1993, Stauffer et al. 2000
Loss of large woody debris	Reduces substrate for feeding, attachment, and cover; causes loss of sediment and organic material storage; reduces energy dissipation; alters flow hydraulics and therefore distribution of habitats; reduces bank stability; influences invertebrate and fish diversity and community function	Ehrman & Lamberti 1992, Gurnell et al. 1995, Johnson et al. 2003, Maridet et al. 1995, Stauffer et al. 2000
Afforestation	Drainage patters, enhanced deposition through scavenging, base cation depletion from uptake and loss through harvesting	Mayer & Ullrich, 1977, Miller 1981, Monteith & Evans 2000, Larssen, T. & Holme, J., (2006)

Lakes

A range of driver-pressure scenarios are known to operate in especially eutrophic lakes (Table 2). The control of primary and secondary pressures arising from these drivers is more commonly conducted. For example, phosphorus (P)-stripping measures in waste water treatment works and reduced nutrient inputs from agricultural fertilisers are now common across Europe leading to a marked reduction in catchment P loading to eutrophic lakes in developed countries (mainly through improvements in sewage treatment and agricultural practices), although, fertiliser applications and P loading in developing countries continues to rise (Withers & Haygarth 2007). For acidification, international agreements and actions to protect and restore natural resources threatened by acidification have resulted in marked reductions in emissions and deposition of acidifying compounds across western Europe and eastern North America (Stoddard et al. 1999).

Table 2. Drivers and pressures checklist for nutrient enriched freshwater lakes that were used in the analysis of pressure-impact relationships. Drivers are underlined and pressures are in italics.

Eutrophication drivers and pressures		
Primary		**Secondary pressures**
Agricultural intensification	*Garden waste*	*Sediment dredging*
Fertiliser run-off	*Fertiliser run-off from gardens*	*Sediments disturbance by boats*
Animal waste run-off		*Metal pollution from mining*
Soil erosion and losses	*Inputs from feeding/roosting waterfowl*	*Invasion by non-native species*
Industrial intensification		*Pesticide discharges*
Textiles discharges	Population growth	*Climate change*
Food manufacturing dicharges	*Sewage discharges*	*Fish stocking*
Paper mill discharges	*Waste disposal*	*Fish removal*
Mining discharges	*Construction discharges*	*Acidification*
Distillery discharges	*Transport/road run-off*	*Macrophyte harvesting*
Aquaculture discharges	*Detergent and soap discharges*	*Water level management*
Tourism and recreation		*Waterfowl introduction*
Food waste	Other pressures	*Extreme weather events*
Fish stocking	*Atmospheric deposition*	*Industrial thermal-regulation inputs*
Sediment disturbance by boats	*Internal nutrient loading*	
	Cyanobacterial N_2-fixation	

Estuarine and coastal waters

Borja et al. (2010) provided an overview of pressures in the estuarine and coastal ecosystems reported in a number of case studies and related those to the different organism groups (Table 3). In total 25 stressors were detected in the literature (Table 3). The most common stressors dealt by restoration were eutrophication and wastewater discharge (24%) and dredging, sediment disposal and sand extraction (22%).

Table 3. Pressures in the marine environment (from Borja et al. 2010).

	Number of case studies		
Pressure	**Benthic invertebrates**	**Fishes**	**Macroalgae & seagrasses**
Eutrophication/Pollution by organic matter			
Oxygen depletion	1		
Wastewater discharge	6	1	1
Sewage sludge disposal	3		
Mine tailings	2		
Fish farm	3		
Pulp mill	1		
Eutrophication	3		1
TBT	1		
Oil-refinery discharge	2	1	
Hydrological-morphological modification			
Land claim	1		1
Sediment disposal	2	1	1
Dyke and marina construction	1	1	
Dredging	3	1	1
Realignment of coastal defences	1		1
Physical disturbance	1		
Fish trawling	1	1	
Lagoon isolation	1		
Marsh and tidal restoration	1	1	2
Sand extraction	1		
Aggregate dredging	1		

Conceptual models of cause-effect chains following restoration in marine ecosystems are lacking in peer-reviewed literature. In some cases Driver-Pressure-State-Impact-Response chains describing degradation of marine waters are available (e.g. Pironne et al. 2005; Figure 1). Most DPSIR-chains deal with the influence of legislation on marine ecosystems and are very general (no empirical evidence).

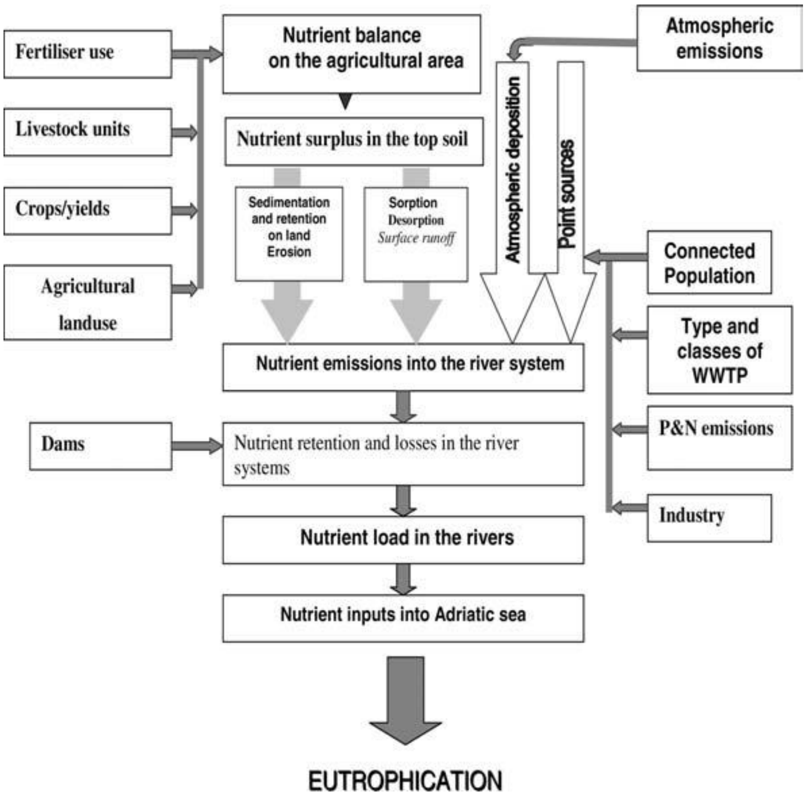
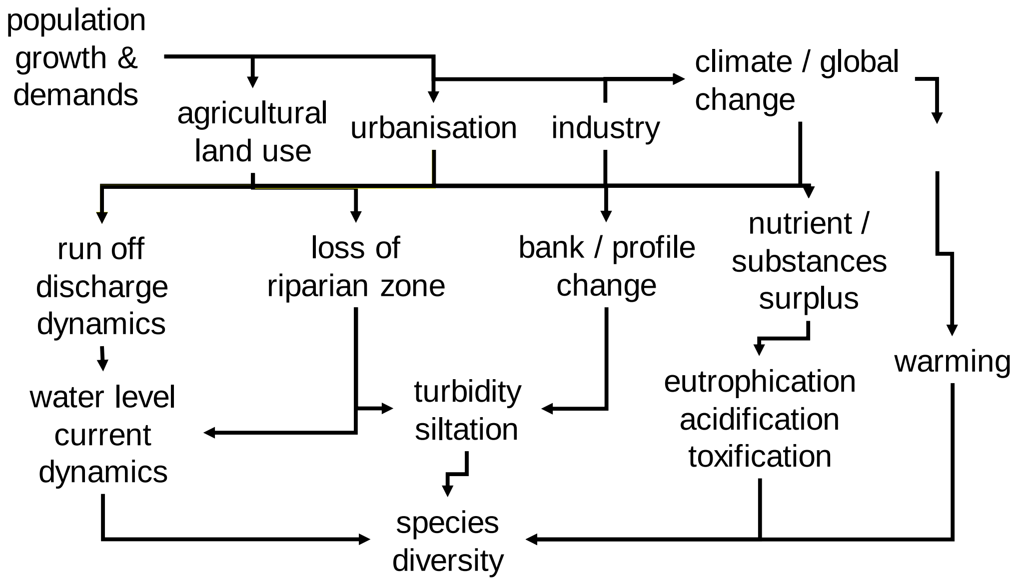


Figure 1. The Driver-Pressure-State-Impact-Response (DPSIR) approach for integrated catchment-coastal zone management: Integrated qualitative approach of different pathways affecting the transport of nutrient from point and diffuse sources in the Po catchment (Adriatic Sea coastal zone system) (from Pirrone 2005).

Summary

Rivers integrate the adverse effects of various activities on land and are, therefore, often simultaneously affected by multiple stressors arising from agriculture, deforestation/afforestation, urbanization, storm water treatment, flow regulation and water abstraction (e.g. Palmer et al. 2010). Globally, lake ecosystems are mainly being affected by eutrophication (intensive agricultural land use) and physical habitat modification of their shoreline, although regionally acidification is still considered as a major threat affecting the biodiversity and ecosystem function (e.g. Johnson et al. 2003). While estuaries and wetlands constitute the ultimate sink for nutrients and other sources of pollution and contaminants originating from entire river basins (Cloern 2001, Diaz and Rosenberg 2008), many estuarine and coastal waters are being physically modified, for instance, for flood protection purposes (e.g., Pollard and Hannan 1994) and navigation (e.g., van der Wal et al. 2002). The conceptual models (DPSIR-chains) of the different water categories are hard to compare. Striking is the difference in the level of detail between rivers and lakes (high) on the one hand and the marine ecosystems (low) on the other. This difference probably has to do with the scale of degradation in rivers and lakes, where it is easier to find/deduct pathways of ecosystem response.

In general, the degradation scheme for all water categories looks as follows:



There is a hierarchy in scale of degradation going from global/supra-regional level of world population growth, climate change and drivers and pressures of land use, urbanisation and industrial development, to (sub)catchment scale were run-off, riparian zones, longitudinal profiles and substances flows are decided, to finally the scale of the lake, stretch or habitat were in-stream, in-lake and in-sea processes operate and organisms find their niches.

Implementing restoration measures implies having knowledge of this hierarchy. At high scale level only legislation measures can be taken, though these tackle the problems at their core source! At the (sub)catchment level both source and effect oriented measures as well as regional legislation can be effective. Measures here are mostly external from the individual water ecosystem. At the lowest local scale level only effect oriented measures are taken. Such measures operate in the aquatic environment itself and are as such internal measures.

Chapter 2. Recovery: Concepts

Rivers

The Conceptual Models presented by Feld et al. (2011) are the most sophisticated ones up to date for streams and rivers. These models show all relationships of management (Response) measures, its effects on different environmental State variables and eventually the Recovery of in-stream organisms (Figure 2, 3 and 4). Feld et al. (2011), however, stressed that no reference in the literature provided statistically significant evidence of an entire cause-effect chain from the Response measure via one or several environmental States to the biological Impact, and this highlights an obvious gap in the currently piecemeal, rather than holistic, understanding that clearly needs to be addressed in future studies. The majority of studies were limited to measuring environmental effects of a Response measure, while comparatively few studies measured the biological Recovery. Moreover, studies on biological effects often missed to attribute their findings to changing environmental States, which renders the construction of cause-effect relationships from these studies difficult.

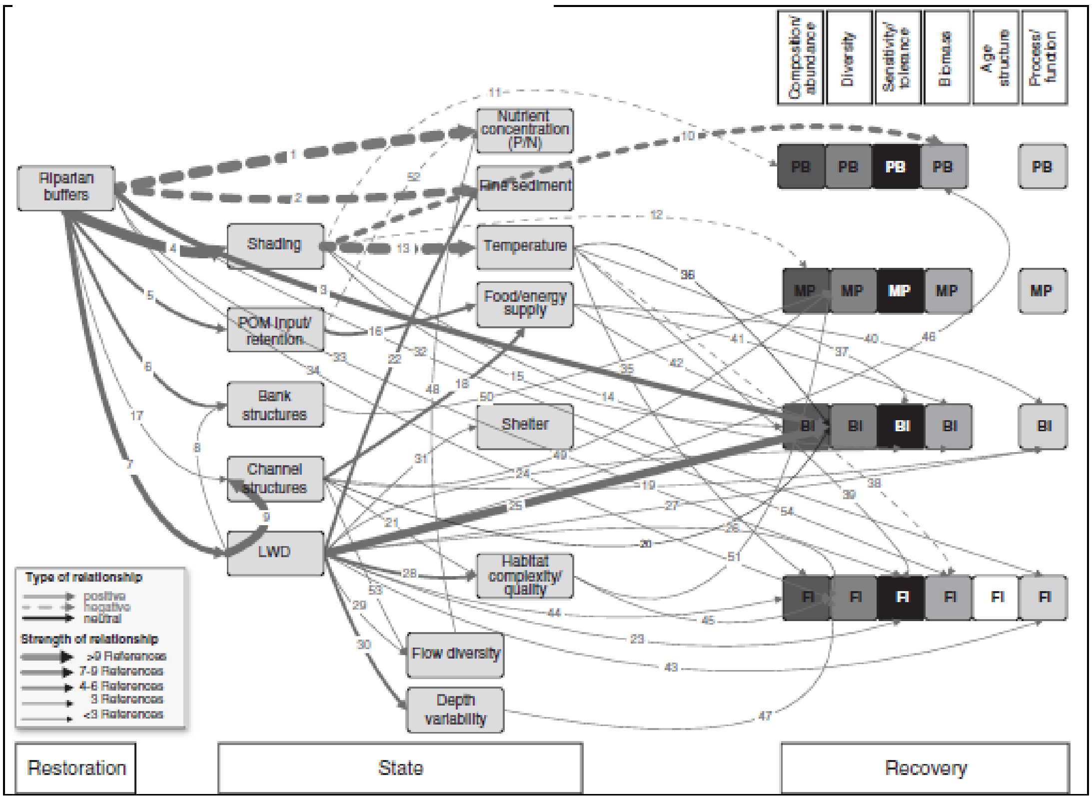


Figure 2. Conceptual model of the effects of riparian buffer restoration as reported in the restoration literature. Boxes on the right 3 represent benthic algae (PB), aquatic macrophytes (MP), benthic macroinvertebrates (MP) and fish (FI). Arrow numbers allow the relation to the references listed in Feld et al. (2011) Annex 2.1.

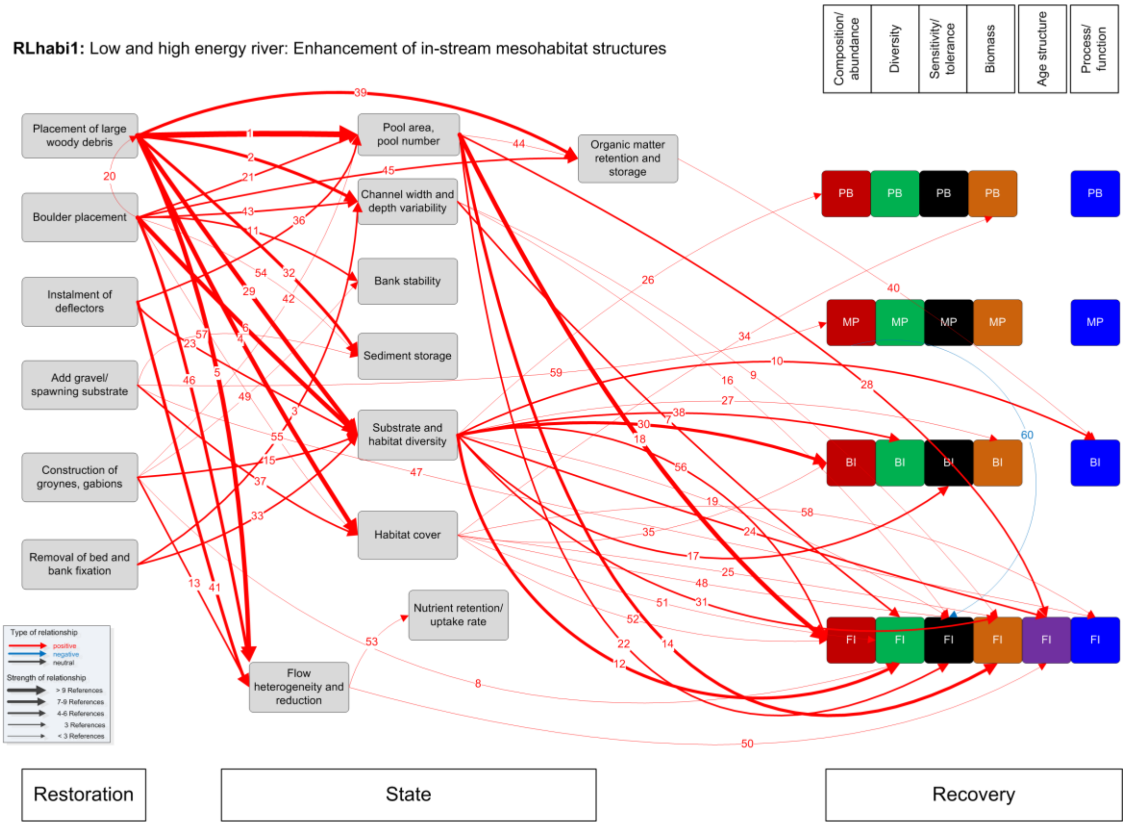


Figure 3. Conceptual model of the effects of in-stream mesohabitat enhancement as reported in the 1 restoration literature. Boxes on the right represent benthic algae (PB), aquatic macrophytes (MP), benthic macroinvertebrates (MP) and fish (FI). Arrow numbers allow the relation to the references listed in Feld et al. (2011) Annex 2.2.

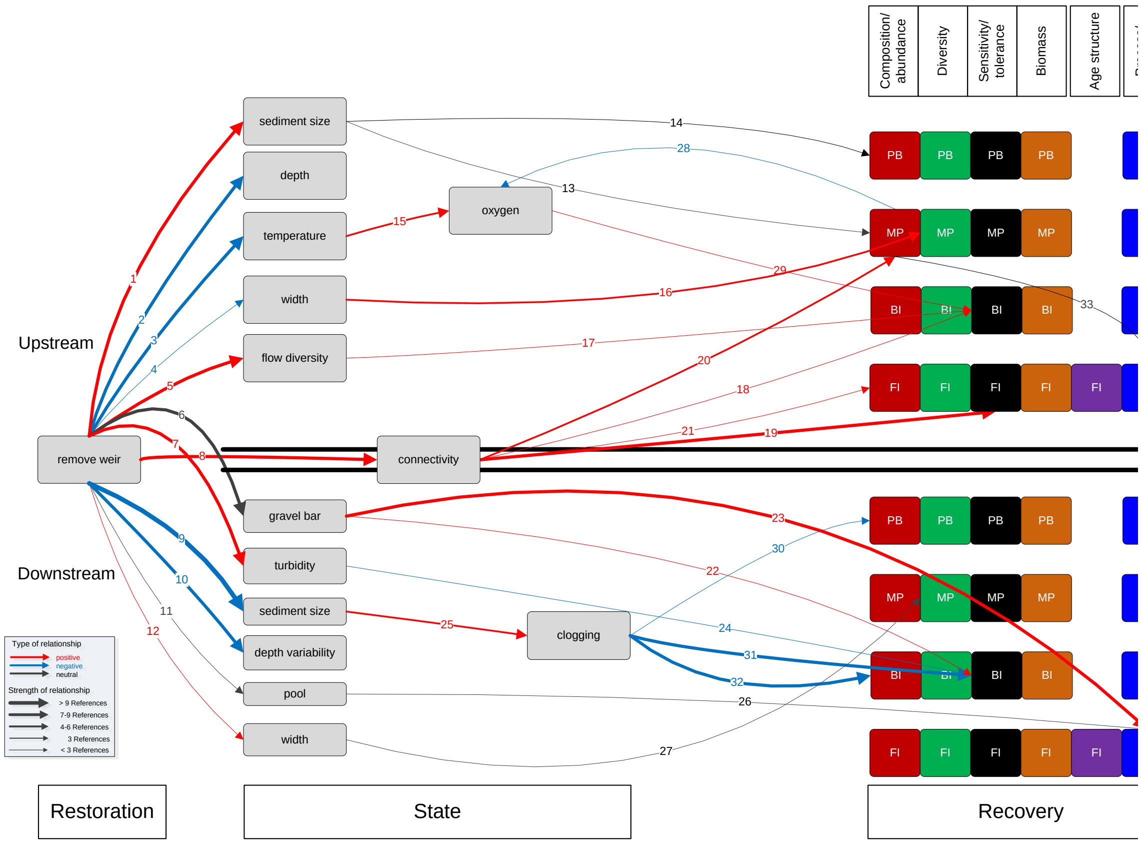


Figure 4. Conceptual model of the effects of weir removal as reported in the restoration literature. Boxes 1 on the right represent benthic algae (PB), aquatic macrophytes (MP), benthic macroinvertebrates (MP) and fish (FI). Arrow numbers allow the relation to the references listed in Feld et al. (2011) Annex 2.3.

Lakes

Thirty peer-reviewed publications, reporting on the management of 41 eutrophic lakes, were evaluated. Three lakes were included more than once in the literature (maximum of 3 occurrences) resulting in 46 lake equivalent case studies. The publications were not selected randomly. Instead effort was taken to ensure that at least 3 publications were reported for a range of eutrophication management measures. These pre-defined management measures, along with the number and percentage of lakes for which data on each measure was reported in the meta-dataset, are reported. The lakes the evaluation is based upon stem from 9 countries and are dominated by very shallow (56 % lakes < 3 m mean depth) and shallow (41.3 % of lakes 3-15 m mean depth) lake types. By far the most common management objective reported in lakes was the reduction of in-lake phosphorus concentrations in an attempt to reduce phytoplankton biomass, either through the control of P loading from the catchment or through controlling internal P cycling (i.e. P capping, hypolimnetic aeration and sediment dredging). However, other management objectives included the control of grazer communities to reduce algal biomass through trophic cascades (e.g. biomanipulation), increasing the resilience of lakes to eutrophication pressure effects by increasing habitat diversity (e.g. installing artificial habitats), and enhancing biodiversity and system resilience through the introduction of desirable organisms (e.g. macrophytes and zooplankton).

Additionally, control of the symptoms of nutrient enrichment (e.g. hypolimnetic aeration; biomanipulation) is commonly conducted, especially when drivers or primary pressures cannot be manipulated. In a recent review of 40 years of literature at Loch Leven, a shallow lake in Scotland, a wide range of feedbacks were identified that demonstrate the diversity of drivers, pressures and changes in ecological state associated with the control of various primary and secondary eutrophication pressures (Figure 5). This is a suited example for a more general lake conceptual model.

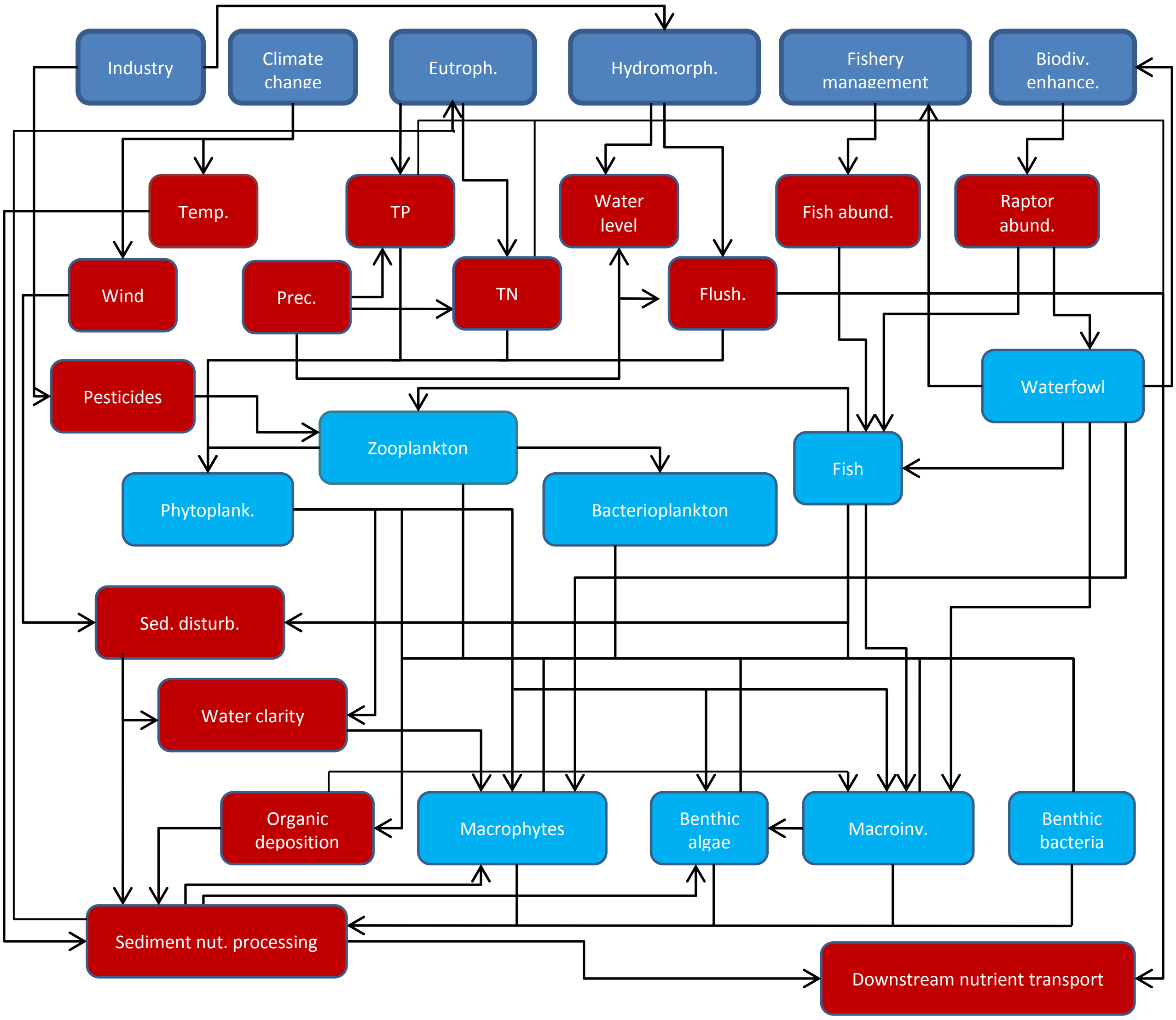


Figure 5. Inter-relationships reported/hypothesised in the literature between primary (eutrophication) and secondary pressures [dark blue], changes to environmental state [red] and impacts on biological quality elements (BQE) [light blue] in Loch Leven between 1968 and 2010.

A review of 30 published manuscripts was undertaken focusing on biological recovery at previously acidified lakes and streams. Of the 30 papers, 21 were concerned with lakes only, 7 focused on streams and 2 covered both lakes and stream sites. A total of 419 lakes and 141 streams were included in the 30 publications although numerous papers featured the same sites. Many papers reported on the results of regional surveys of large numbers of sites while a number of single site studies were also included. The majority of papers papers assessed 'natural recovery' following reductions in atmospheric depsostion of acidifying compounds but several focused on the outcome of liming manipulations. Liming is commonly used in some regions (e.g. Sweden) to mitigate the effects of acidification on biodiversity and ecosystem function of poorly buffered lakes. Liming of sites directly affected by acid deposition is commonly done (e.g. using a helicopter or boat), however some measures have also included liming of wetlands adjacent to lakes and over-liming of lakes to treat watercourse originating as lake outflows. Most managers see liming as a short-term, mitigating measure, although liming has been ongoing for more than 30 years in some regions. International measures agreed upon some 20 years ago have been associated with improved water chemistry and to some extent recovery of lake biology.

Estuarine and coastal waters

Borja et al. (2010) provide a synthesis on the knowledge of recovery patterns in marine ecosystems (based on 51 studies), which it should be possible to construct conceptual models (Table 4).

Table 4. Summary of knowledge on recovery patterns in marine ecosystems (based on 51 studies) (from Borja et al. 2010).

Mechanisms for recovery	**Recovery features**	**Recovery**
From sediment modification	Usually in areas of high sediment turnover and sediment influx, with or without organisms colonising	A function of the ease of sediment influx and the organism influx
By habitat creation	Create the appropriate physical environment and then allow organisms to colonise	A function of the ease of creating the
By organic matter degradation and reduction of nutrient load	Recovery occurs once the excess organic matter is broken down (in the case of sewage and oil), any toxic pollutants have evaporated (from oil spills), and the excess of nutrients is removed; this is more difficult in fine sediments than coarse sediments and in low-energy areas than in high energy areas	A function of the original amount of organic matter stored in the system and the conditions for its breakdown; shown by an absence of symptoms of eutrophication (algal blooms, oxygen depletion, etc.)
From persistent pollutants	The ability of the system to sequester/bury the persistent pollutants or disperse them to reach low background levels	A function of the original amount and toxicity of the pollutants, their degradation potential by physical, chemical or biological methods and thus the speed of sequestration

From excessive biological removal	The ability of the system either to replenish stocks naturally or with human interference through restocking	A function of the severity of the biological removal (overfishing) and the rate of recolonisation/recruitment and reproduction
From hydrological–morphological modification	The ability to remove barriers and restore water flow, current patterns, salinity balance, etc.	A function of the ease with which these hydromorphological conditions can be restored naturally or with human interventions

Summary

The Driver-Pressure-State-Impact-Response-Recovery (DPSIRR) scheme provides a framework that links socio-economy with ecology (EEA 2007, Feld et al. 2011). Literature was searched for existing DPSIRR-chains for the three water categories. Such conceptual models on the recovery of river, lake and estuarine and coastal ecosystems were scarce and fragmented. Such models lacked for the marine systems were quite one-sided, focusing on eutrophication, for lakes and quite specific for certain measures in rivers. Comparison and integration of DPSIRR-chains is until today quite impossible.

Still, more in general some lessons can be extracted:

- Degradation ≠ Restoration
- Knowledge progress is limited and is mainly qualitative descriptive
- Little quantified knowledge appeared available on recovery processes
- (Some) knowledge can be extracted from degradation processes

What at least is needed at this stage of restoration ecology are:

- Well designed BACI monitoring of restoration (few stressors)
- Monitoring over a long time after
- Quantified knowledge on thresholds

Chapter 3. Recovery: Measures

Rivers

Feld et al. (2011) reviewed existing knowledge regarding three frequent types of restoration measures including re-establishment of riparian buffer strips, in-stream habitat improvements and weir removal. Water quality improvement by riparian buffers primarily aims at buffering the adverse impacts of intensive agricultural land use adjacent to streams and rivers. A sufficiently wide and ideally mixed riparian vegetation strip at both sides of a stream is considered to retain plant nutrients (e.g., nitrogen and phosphorous components), fine sediments and toxic substances (e.g., pesticides) that enter streams via surface runoff from adjacent agricultural areas (Barton et al. 1985; Castelle et al. 1994). Riparian trees provide shade and organic material (leaf litter, wood) and thus food and shelter for in-stream biota (Parkyn et al. 2005; Davies-Colley and Quinn 1998; Davies-Colley et al. 2009).

The enhancement of in-stream mesohabitat structures aims at increasing structural diversity and is often considered to promote biological diversity (Palmer et al. 2010). In particular the introduction (or omission of the removal) of LWD provides a key habitat for fish and benthic macroinvertebrates (Roni and Quinn 2001; Kail et al. 2007) and also stimulates habitat diversity (e.g., creation of pools) by diversifying hydraulic conditions (Baillie et al. 2008). Besides LWD, we evaluated the mitigation effects of the introduction of boulders, deflectors, fish spawning substrates and the removal of bank enforcement (e.g., sheet piling or rip-rap).

The removal of weirs and dams primarily aims at restoring the longitudinal connectivity of streams and rivers. Weir removal is considered to promote the migration of fish and benthic macroinvertebrates (Gregory et al. 2002; Doyle et al. 2005), and there are effects on flow conditions and sediment particle size upstream and water temperature up- and downstream (Bednarek 2001; Hart et al. 2002).

Another more common technique for rehabilitating rivers and reconnecting them to their floodplains is to construct a new meandering channel (Roni et al. 2005). This intervention has been used frequently in rehabilitation projects in the lowlands of the European plain. As a result, re-meandering interventions are also seen important in rivers.

Lakes

The state-of-the-art reviews on restoration of shallow, eutrophic lakes in Europe (Gulati and Van Donk, 2002; Søndergaard et al., 2007) have drawn several generalisations about the progress of lake rehabilitation works in NW Europe and all focus around biomanipulation. These reviews are based on numerous studies on temperate lakes carried out mostly in Denmark, the Netherlands, Germany and Sweden (Gulati et al., 1990; Lammens et al., 1990; Jeppesen et al., 1990; Hansson et al., 1998; Meijer et al., 1999; Benndorf et al., 2002; Mehner et al., 2002; Van de Bund and Van Donk, 2002; Jeppesen et al., 2005). Many of these studies covered both whole-lake, enclosure and laboratory scale experiments and advanced our knowledge of theory and mechanisms behind the food-chain processes.

Søndergaard et al., 2007 evaluated data from more than 70 restoration projects conducted mainly in shallow, eutrophic lakes in Denmark and the Netherlands. Special focus was again given to biomanipulation, the removal of zooplanktivorous and benthivorous fish, by far the most common internal lake measure.

In the 46 lake equivalent case studies a range of single (63%) and combined (37%) management measures were reported (Figure 6).

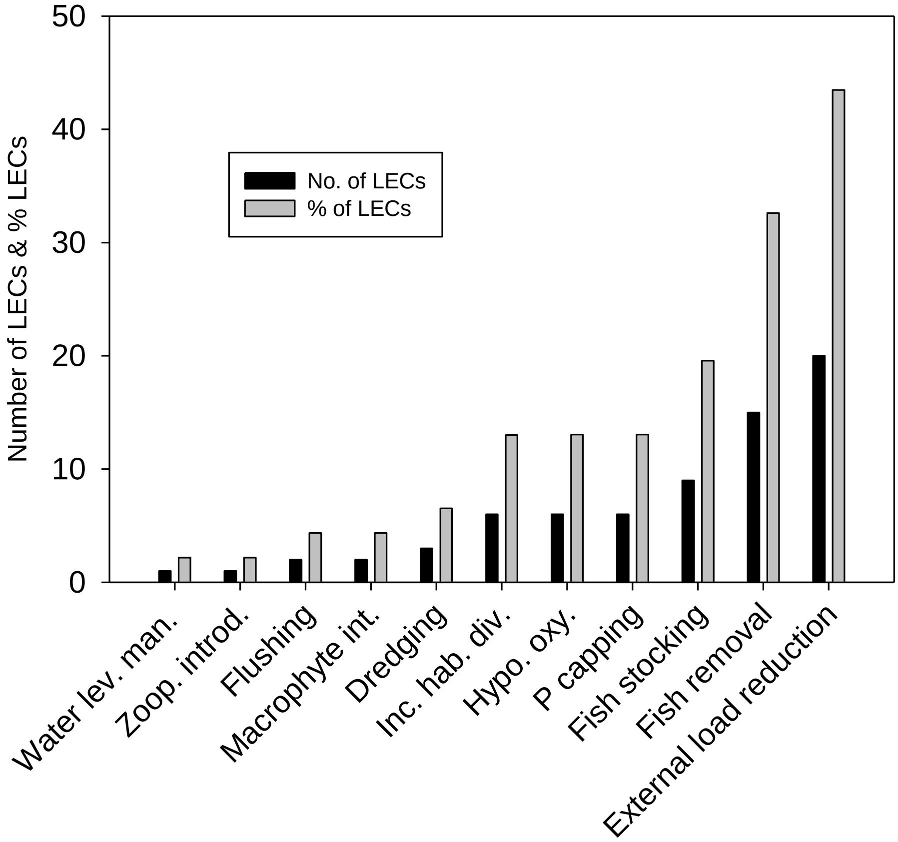


Figure 6. Summary of management measures targeted in the meta-analysis and the distribution of lake equivalent case studies (LECs) in which each of the measures was conducted.

The use of combined management approaches was also conducted both consecutively and simultaneously. However, reductions in in-lake P concentrations were commonly reported, although the wide range of pre- and post-management annual mean TP concentrations estimated from reported data was large, the latter commonly well exceeding WFD lake type specific TP targets (Figure 5). Only 11% of lakes reported post-management annual mean TP concentrations below 0.1 mg TP l^{-1}. The range of pre- and post-management annual mean TP concentrations reported for catchment nutrient load reduction alone (n = 7) were 0.1 to 1.6 mg TP l^{-1} and 0.03 to 0.25 mg TP l^{-1}, respectively (delta TP range 1.35 to 0.05 mg TP l^{-1}). For this management measure, only 4 lakes reported complete recovery in annual mean TP concentration with transient recovery periods estimated between 1 and 15 years. It is highly likely that the confidence associated with such estimates is low due to the low number of lakes reporting both single management measure scenarios and resulting changes in TP in the meta-dataset. The only other management measure with more than 3 lakes reporting pre- and post-management TP concentrations (ranges of 0.55 to 0.15 mg TP l^{-1} and 0.73 to 0.08 mg TP l^{-1}, respectively; delta TP range of -0.4 to 0.1 mg TP l^{-1}) was biomanipulation of fish stocks by removal (n = 5) where effects appeared to be highly variable. An analysis of variance between the pre-management and post-management TP concentration data populations, using non-parametric Kruksal-Wallis test, indicated post-management TP concentrations were significantly lower than pre-management TP concentrations ($p < 0.01$).

Liming of lakes and watercourse began in earnest in the early 1980s in Sweden. Henriksen and Brodin (1995) edited a book summarizing about 20 years of experience of lake liming practices in Sweden. A number of different liming strategies have been used, such as the liming of individual lakes to treatment of whole or large parts of a catchment. For long-term effects, catchment liming practices are considered the best. However, due to the high costs and lag-phase responses from slow processes of soil neutralization, this method is often considered too expensive by managers. As an alternative, liming of wetlands is considered the second-best strategy. A problem with this type of management was the effects that liming may have on wetland fauna and flora. In contrast to whole catchment and wetland liming, lake liming is comparatively inexpensive, but this management strategy requires frequent treatments to alleviate the episodic effects of acidic water from snowmelt or heavy rainfall. Dosage concentrations are adjusted for water pH and lake retention time, which often results in 10-75 g/m^3 lake water. It is recommended that treatments be repeated periodically to keep water pH above 6.0. For wetland liming, 10-30 g of limestone per m^3 has been recommended. In Sweden, some 7000 lakes and 11 000 km of watercourses are limed at a yearly cost of ca. 1.8 million €, in order to restore biodiversity (i.e. facilitate the recovery of acid sensitive biota) and create conditions for recreational fishing (i.e. protect and enhance existing fish populations; Appelberg & Svensson 2001; SEPA 2007).

Estuarine and coastal waters

A clear overview of measures taken in the coastal and estuarine environments lacks. Most measures deal with inland legislation.

Summary

In rivers most measures target the morphology of the stream stretch (mainly remeandering) or the instream habitats. Few only are related to reduction of nutrient input (Figure 7).

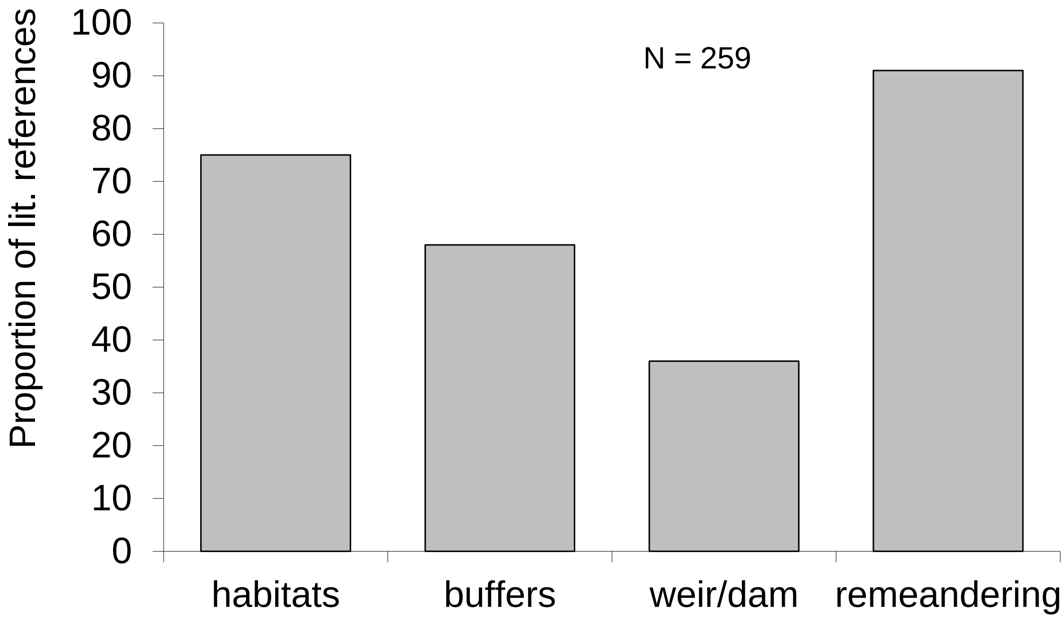


Figure 7. Proportion of literature references related to hydromorphological restoration measures taken in rivers.

In acidified rivers out of 141 sites 131 endured external pressures and at 8 sites liming is considered as a secondary measure as it is underaken within the context of reduced amospheric depsostions.

On the contrary, in lakes all measures target to reduce nutrient levels, especially phosphate (Figure 8). Others mainly focus on acidification (out 419 sites 371 endured external pressures and 2 toxic substances).

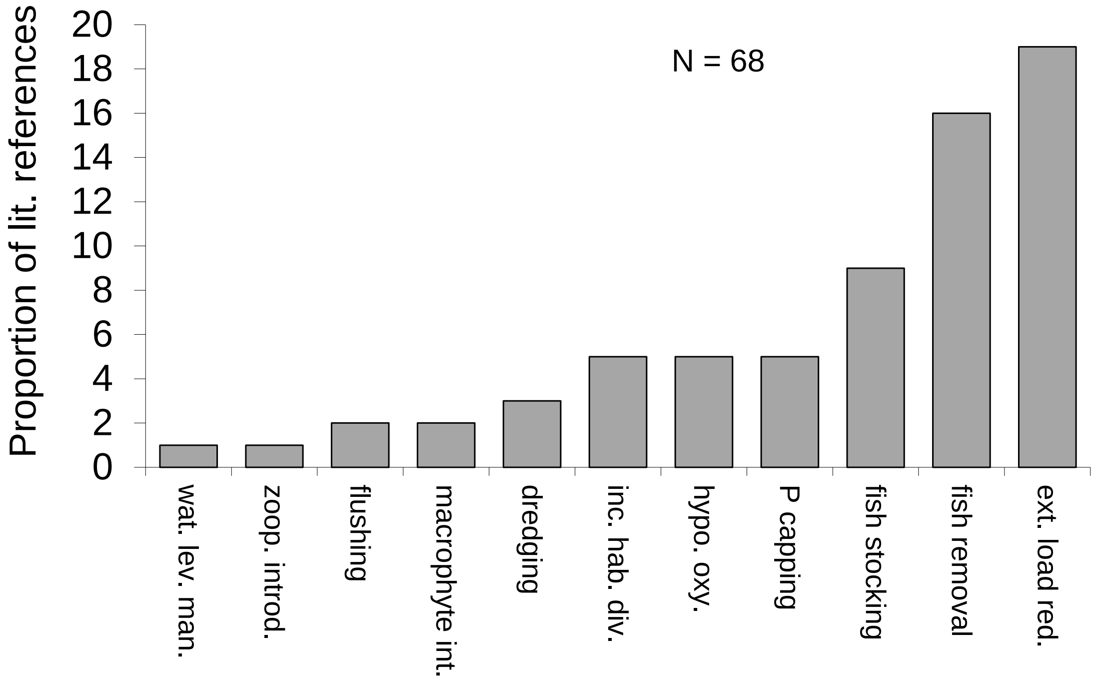


Figure 8. Proportion of literature references on additional measures included in nutrient reduction projects in lakes.

Measures are not often taken directly in estuarine and coastal waters, these much more relate to measures taken inland through legislation on nutrient reduction (Figure 9).

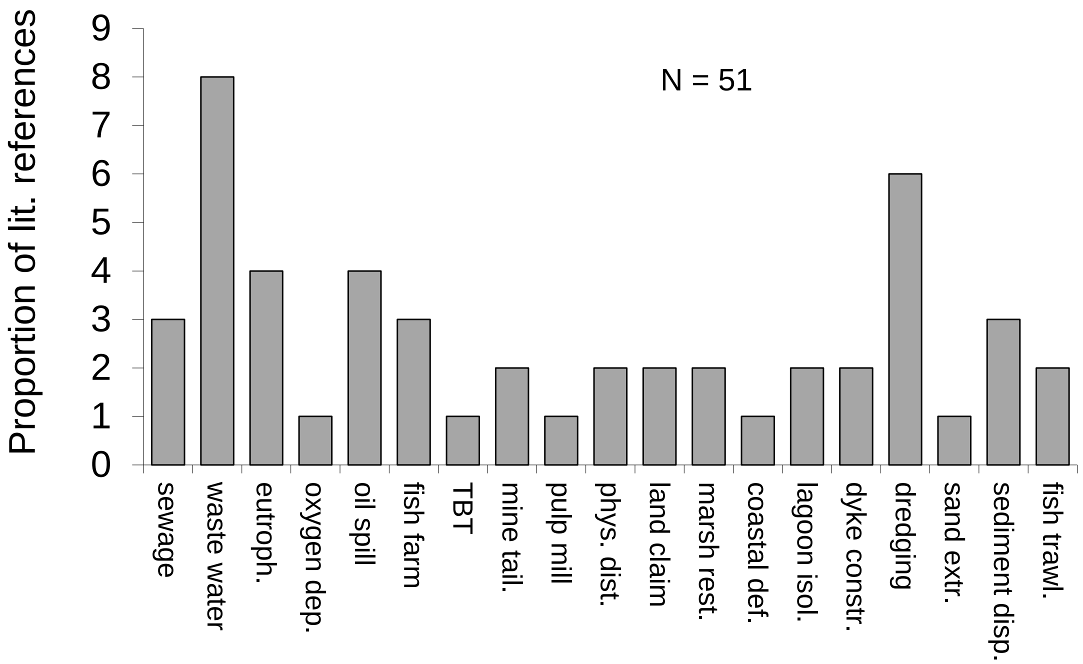


Figure 9. Proportion of literature references related to restoration measures taken in estuarine and coastal waters.

These observations support our initial hypothesis that “at a catchment scale, nutrient stress affecting functional (production/decomposition) processes will be more important in lakes and marine systems, while hydromorphological stress affecting habitat availability will be more important in rivers”. Surprisingly little information on other stressors is available but these hamper full recovery!

Chapter 4. Recovery: Data availability and processing

Rivers

Although, river restoration is getting increased attention in many parts of the world (Palmer et al. 2007), the lack of proper monitoring data seems to be a general problem. Bernard et al. (2005) stated that of 37.000 river restoration projects in the United States, only 10% included some form of monitoring and of this information little was either appropriate or available. Feld et al. (2006) conducted a study on the effectiveness of several hundreds of ecological river restoration projects in Germany. They found that for less than a quarter of the studies (23%), post-project evaluation of measures had been conducted (Keizer-Vlek et al. 2011).

An overview of European papers on river restoration, studied by Reitberger et al. (2010), showed that none of the studies did look at time series. They all reflected cases of control-impact studies. In most case space-for-time substitution is used to study recovery times; although only 8 years after restoration was the longest monitored series. In most studies control-impact design was common. In some studies groups of streams with different times after restoration were used (another form of space-for-time substitution). Striking was that almost no pre-restoration data were available. Some exceptions are given in a study by Louhi et al. (2011).

Lakes

The general data analysis approaches employed in the 46 lake equivalent case studies are summarised (Figure 10). The majority of the lakes employed ‘before and after’ and ‘time series’ approaches. The majority of lakes employed multiple data analysis approaches (4 lakes used 1 approach; 34 lakes used 2 approaches; and 8 lakes used 3 approaches). The use of statistical analyses to validate the data analysis approaches across the 46 lakes is summarised (Figure 11). The majority of the 21 lakes in which statistical analyses were used to test some aspect of the recovery process used ANOVA or regression analyses. 26 % of the 46 lakes considered longer-term recovery effects (i.e. > 10 years; Figure 12).

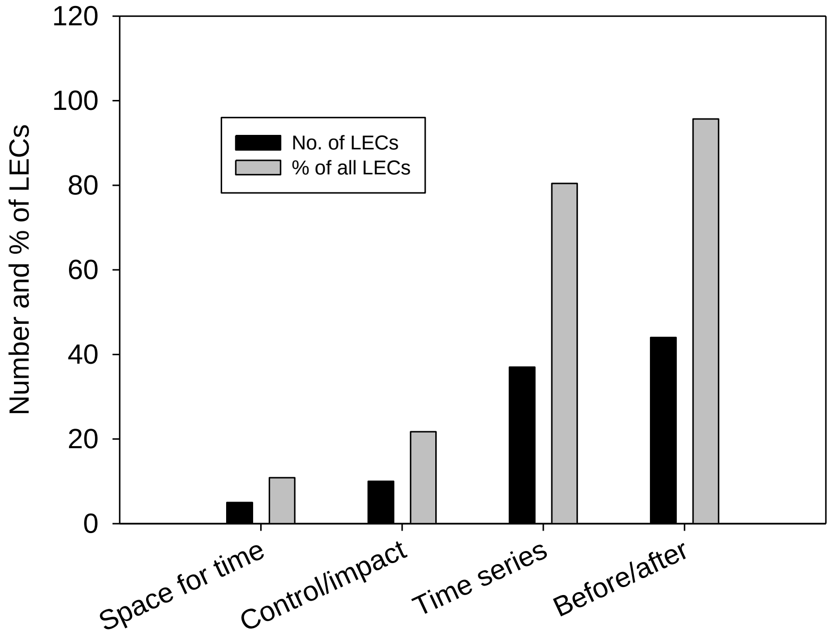


Figure 10. Number and proportion of 46 lake equivalent case studiesin which space for time, control/impact, time-series and before/after data analysis approaches were used.

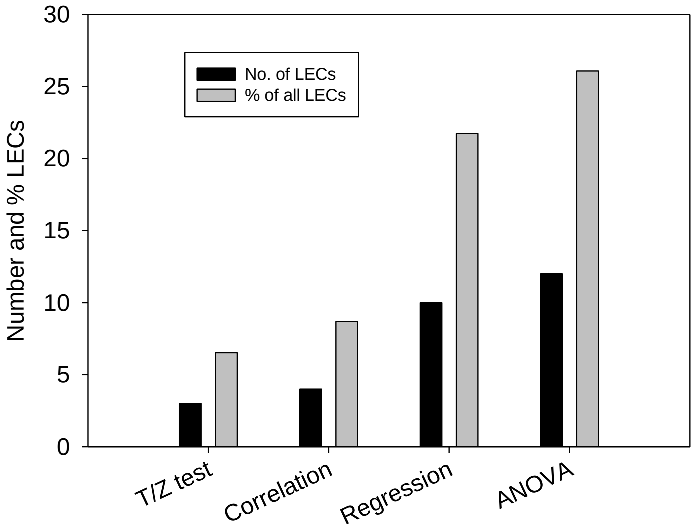


Figure 11. Number and percent of 46 lake equivalent case studies in which statistical analyses were conducted to test the effects of management measure scenarios.

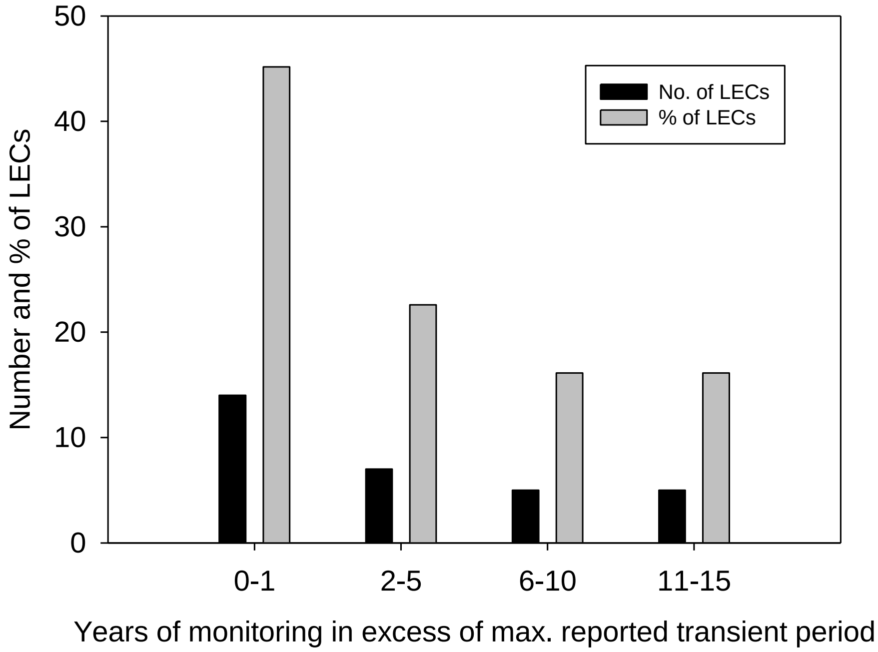


Figure 12. Distribution of lake equivalent case studies (n = 36) for which monitoring was conducted in excess of the estimated maximum transient recovery period.

The majority of lake liming studies lack robust pre-liming conditions. Indeed, most liming projects were started simply if lake water pH dropped below 6.0. Knowledge of biological conditions was seldom measured as were other water chemistry variables. Analyses to determine the effects of liming are generally based on time-series data of surface water chemistry and comparison of lake biology with nearby reference lakes. Palaeolimnology has proved an important means of assessing the timing, extent and causes of acidification since the mid-ninteenth century. In particular the development of the diatom-pH transfer function and the use of sub-fossil material from lake sediments to identify pollution from long-range sources have provided valauble information in the absence of historical / monitoring data over this period. Palaeolimnological data can now be compared with observational time series data from monitored sites to provide a means of validating the sub-fossil based reconstructions. Diatoms and the remains of other organism groups stored in lakes sediments have been used to track both degradation and recovery phases.

Estuarine and coastal waters

Data availability and processing is low in estuarine and coastal waters (Borja et al. 2006).

Summary

In rivers and lakes quite an amount of monitoring data are available. In estuarine and coastal waters such data are scarce. Despite the number of monitored recovery cases, each one seems to stand alone as monitoring schemes were set-up for local situations and to answer partial questions. Furthermore, in many, many cases data on recovery just lack and this is quite alarming! Not only is the amount of available data surprisingly low, the composition of the available data is often very limited and does not allow the evaluation and generalisations of improvements and eventually of successes. This can be due to several reasons. First, an overwhelming majority of restoration measures is not being monitored at all, probably because there is no legal requirement that designates restoration monitoring mandatory. And second, if restoration measures are monitored, the methods and time-scales applied rarely fit the state-of-the-art in freshwater monitoring but are based on common state and trend monitoring. Third, most water authorities do not focus on long term ecological success but focus on getting the job done, thus taking the measures intended.

Before-After-Control-Impact monitoring would be the only best option to monitor recovery. As can be seen this was only done in a limited percentage of river restoration studies (solely experiment). Most focus on Before-After approaches. Time series lack in rivers but are available for lakes. The marine monitoring is less well described.

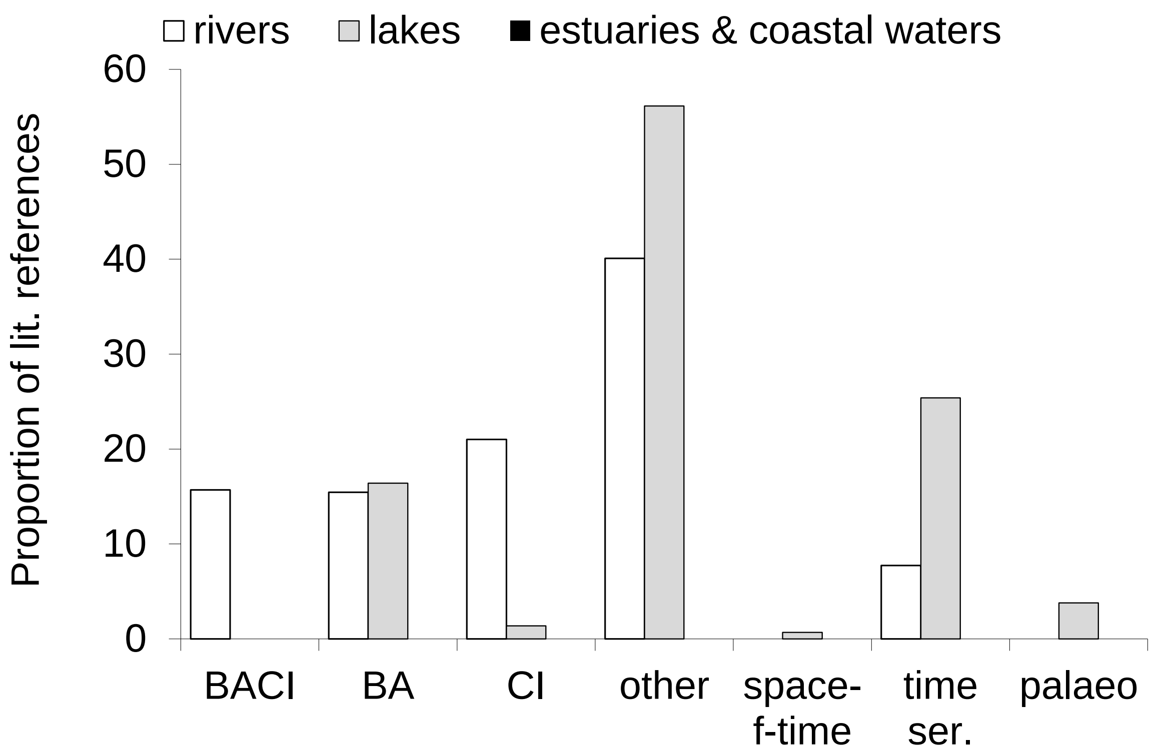


Figure 13. Proportion of literature references that refer to specific data evaluation techniques as applied in river, lake, estuary and coastal water restoration projects.

The huge investments in recovery of surface waters require control of the ecological effects. Therefore, restoration monitoring should become mandatory. Only by frequent monitoring of biological and abiotic changes after restoration will restoration practitioners and scientist be able to evaluate the success of the restoration measure and eventually of the investment done.

Restoration monitoring requires a tailor made sampling design that allows of sound statistical analysis according to state-of-the-art methods. First, in order to monitor changes, the status before restoration must be recorded at least once. Second, the status after restoration must be recorded several times in order to account for the development of a restored site after restoration. And third, a control (non-restored) site similar to the restored site before restoration must be monitored in order to detect the effect of natural variability (and climate change) and subtract them form pure restoration effects. This before-after-control-impact (BACI) design is standard in scientific research and allows the statistical testing of restoration effects and recovery.

Chapter 5. Recovery: Successes

Rivers

Feld et al. (2011) conclude that: “The huge body of restoration literature verifies that restoration does show effects on both the environmental variables that make up aquatic habitats and the biological communities that recover and occupy these habitats. Riparian vegetation can effectively buffer nutrients and sediments with various positive effects on the in-stream fauna and flora. The placement of (natural) in-stream habitat structures does lead to measurable effects on the in-stream fauna. Finally, the removal of weirs and other small barriers does restore the longitudinal connectivity and habitat diversity upstream.” This conclusion by Feld et al. (2011) is completely opposite to the conclusions drawn by Palmer et al. 2010. Palmer et al. reviewed the results from 78 different restoration projects that quantitatively examined the reach-scale response of invertebrate species richness to restoration actions that increased channel complexity/habitat heterogeneity. In contrast to Feld et al. (2011), they report that while “most projects were successful in enhancing physical habitat heterogeneity; only two showed statistically significant increases in biodiversity rendering them more similar to reference reaches or sites”. On the other hand, Feld et al. (2011) considered all BQEs and more response parameters, thus comparing reviews in this case is flawed by the different targets of the reviews. Besides the study by Palmer there are several other recent studied that did not provide very encouraging results when restored stream channel biological conditions were compared to reference sites (e.g. Jähnig et al. (2010), Louhi et al. (2011), Sundermann et al. (2011) and Violin et al. (2011)).

According to Miller et al. (2010), who published a comprehensive meta-analysis of the effects of mesohabitat enhancement on benthic macroinvertebrates, there is a “myriad of weakly replicated, inconclusive, and even conflicting published studies”. The authors point at some general flaws in restoration science (e.g., lack of sound study design including inappropriate replication, or publication bias) and question the methods to evaluate treatment effects (see also Shields 2003). Study designs lacking pre-restoration data render impacts on macroinvertebrates questionable as these communities vary naturally at small spatial scales. Furthermore, conclusions about restoration significance remain unrelated if only unrestored, but not undisturbed controls are being used to detect the effects (Feld et al. 2011). Apart from this effectiveness or success can only be defined if initial criteria are specified with which the recovering system can be measured against. This may be a historical reference, a comparable river section that is considered to be in a reference state or aims and goals such as a particular sinuosity or the colonization of certain species. A large proportion of projects do not define any criteria or give an indication if the changes seen following restoration interventions indicate progress towards recovery. The concept of success suggests a positive end point where a dynamic equilibrium is achieved in the newly modified system. However, rate limiting factors such as floods, droughts, dispersal will mean that population, community and ecosystem responses to the addition of habitat will often take considerable lengths of time (Lake, 2001). These sorts of delays do not cause restoration to fail, but instead, may push response times beyond those over which monitoring is typically funded (Bond & Lake 2003). Projects were therefore analysed in terms of their progress towards rehabilitation goals rather than their outright 'success'.

An evaluation of the indicators that responded positively to restoration showed that the indicator groups 'river banks' and 'riparian & floodplains' demonstrate the largest positive response to stream restoration (100% and 85%, respectively)(Reitberger et al. 2010). Morphology, macroinvertebrates, diatoms & algae, and bed load showed positive responses on 59%, 57%, and 50% of occasions respectively. Macroinvertebrates, fish, and hydraulics as the most frequently used indicators in this study, scored surprisingly low positive response rates compared to the rarely used indicator groups 'river bank' and 'riparian and floodplains'. Water quality and chemical indicators showed least effects. Unexpectedly, also indicator species mainly failed to show positive response to restoration within the first years after project implementation (Figure 14).

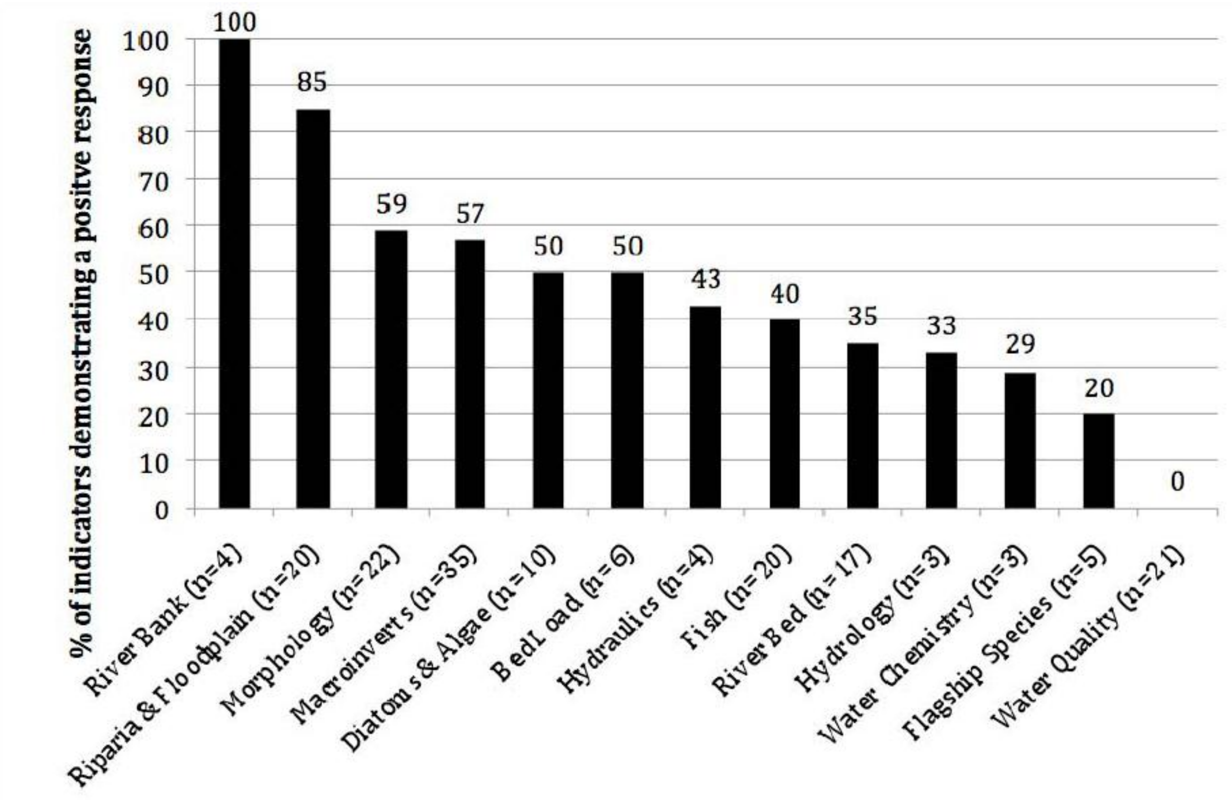


Figure 14. Positive short-term indicator response per indicator group (n=total number of indicators analysed within each group (Reitberger et al. 2010)).

The majority of abiotic indicators applied in the reviewed literature by Reitberger et al. (2010) did not reliably respond to the restoration efforts made (Table 5), which may indicate either that indicators were 'unsuited' or that the measures did not show the desired effects at all, i.e., restoration was not successful. Solely the amount of large woody debris (LWD) following restoration seems to be an unambiguous indicator of improvement. Similarly, changes in channel form or instream habitat are often due directly to restoration actions (manipulations of channel morphology or addition of in-stream structures) and they may not remain intact over the long-term. This is particularly true if the stream reach that was 'restored' exists within a broader watershed context that is degraded (Palmer, 2009). Further, increases in fish abundance that have been found when in-stream structures or LWD are added may simply be due to local aggregations of fish and not an increase in fish production.

Table 5. Indicators of physical structure and water quality and their response to restoration efforts. Indicators reported in >10 studies are indicate in bold. Again, positive response means that the indictor changed post-restoration toward the desired direction. Negative suggests the ecological conditions worsened (with respect to the indicator) and no response means there was not a significant change (after Reitberger et al. 2010).

Indicators				
	Positive response	Negative response	No response	Response unclear
Physical structure				
Bed particle size	2	2	1	
Sedimentation	3	4	**25**	
Bed erosion/scour		3	2	
Bank erosion	**10**	4	**16**	
Width:Depth ratio	8	5	**35**	
# pools/length	3	3		16
'habitat' score	2	8	**17**	
Habitat heterogeneity	7	3	8	
Large woody debris	**11**	1	3	
Temperature		1	2	
Velocity			5	
Light	1	1		
% Organic matter (sediment0	4	1		
Water quality				
pH			1	
DO	1		1	
NO3 or total N	1		2	
DOC				2
SO4				1
Conductivity				1
PO4	2			
Suspended sediment	1	1		

Lakes

In 82% of the lakes in the 46 lake equivalent case studies a reduction in annual mean TP concentration was achieved of the lakes for which pre- and post-management TP concentrations were reported. However, the reported end-point recovery TP concentrations were commonly high (i.e. > 0.3 mg TP l^{-1}) in relation to lake type specific WFD TP targets. Many of the lakes did not, however, provide evidence of recovery of multiple BQEs following TP reduction.

Jeppesen et al. (2005) concluded in an evaluation of 35 long-term data series:

- Summer mean TP concentration declined in 76% of the shallow lakes and in all deep lakes. Reductions in annual mean TP concentration occurred in 86% of the shallow lakes and nearly all deep lakes.
- Summer TN concentrations declined in 83% of the shallow lakes, whereas no consistent pattern was found in N loading reductions in the deep lakes.

- The TN : TP ratio in the lakes was positively related to the TN : TP ratio in the lake inflows, both during summer and annually, and was also positively related to depth. With decreasing TP concentration, the TN : TP ratio increased markedly in both deep and shallow lakes. An increase in the summer TN : TP ratio could be seen in 80% of the lakes receiving water with an increased TN : TP ratio, but the TN : TP ratio even increased in a few lakes for which the TN : TP ratio of the inflowing water decreased.
- In all lakes except one, the summer SRP concentration declined with decreasing TP concentration, while no changes or even increases were found in lakes with no changes or increases in summer TP concentration.
- No clear pattern was observed for DIN in individual lakes. However, the summer DIN : TN ratio increased in 76% of the shallow lakes, and in 82% of the deep lakes.
- In 80% of the shallow and 91% of the deep lakes, the summer DIN : SRP ratio increased with decreasing TP loading and TP concentration in the lake.
- In 71% of the shallow lakes and in 69% of the deep lakes, a decline was found in summer chl a concentration with decreasing summer TP concentrations. These were 76% and 64%, respectively, when chl a values were averaged on an annual basis.
- In 77% of the shallow and 82% of the deep lakes, the Secchi depth increased as nutrient loading decreased.
- In most cases fish responded strongly to the reduction in nutrient loading. In 82% of the lakes with available fish data, decreases were noted in the catch of fish by either commercial fishermen, anglers or in fish surveys.
- The total zooplankton biomass increases with TP concentration and decreases with depth.
- Phytoplankton biomass followed the pattern for chl a concentration; it declined in 71% of the shallow lakes and 70% of the deep lakes. For shallow lakes, the contribution of diatoms to the total biovolume increased in 69% of the lakes, and the contribution of cryptophytes and chrysophytes in 63% and 64% of the lakes, respectively. No significant pattern was found for the remaining phytoplankton groups. The contribution of chrysophytes also increased in 82% of the deep lakes. In addition, an increase in dinophytes was found in 75% and a decline of cyanobacteria in 80% of the cases.
- The response of macrophytes to reductions in nutrient loading was not uniform across lakes. In most lakes for which data were available, signs of macrophyte spread were apparent, either as an increase in macrophyte abundance, coverage, plant volume inhabited and/or, in the case of submerged macrophytes, depth distribution.

Widespread increases in surface-water pH have been attributed to international actions to improve air quality (Stoddard et al. 1999). Since surface-water chemistry exerts a major control on aquatic biodiversity (Resh and Rosenberg 1993), improved surface-water quality (e.g. increased pH) should result in biological recovery, albeit with inherent time lags (Evans et al. 2001). For example, chemical recovery from acidification is characterized by marked increases in pH and alkalinity and decreases in SO_4^{2-} concentration, whereas biological recovery can be characterized by decreased predominance of acid-tolerant taxa and recolonization of acid-sensitive taxa. A growing body of literature documents improvement of surface-water pH. However, records of biological recovery are scarce, and results are equivocal (Skjelkvåle et al. 2000, Alewell et al. 2001, Stendera and Johnson 2008, Angeler and Goedkoop 2010).

Estuarine and coastal waters

Borja et al. (2010) surveyed the current literature and identified 51 long-term estuarine and coastal restoration cases. Most showed recovery after a shorter or longer period of time. In a few cases, recovery was not at all evident. From four well-studied coastal ecosystems, Duarte et al. (2009) did not observe a return in simple biological variables (such as chlorophyll a concentration) following the assumed reduction of nutrient loads during two decades. In the Chesapeake Bay, despite extensive restoration efforts (including point-source reductions, fisheries management, sea grass plantings and oyster bed restoration), nutrient concentrations and associated ecological health-related water quality and biotic metrics have generally shown little improvement and, in some cases, large decreases since 1986 (Williams et al. 2010), keeping the submersed aquatic vegetation coverage below restoration targets (Orth et al. 2010). This may be reflected by the hysteresis term in the model proposed by Elliott et al. (2007) which indicates that the trajectory of degradation may be different from the trajectory of recovery; that difference can be regarded as a degree of ‘memory’ in the system (Peterson 2002) which may be related to the type of stressor and the ability of it to be assimilated.

Summary

In rivers, the environmental improvement (positive response) was on average the case in 33% of the projects (Figure 15). The biological positive response accounted for 50% of all projects evaluated.

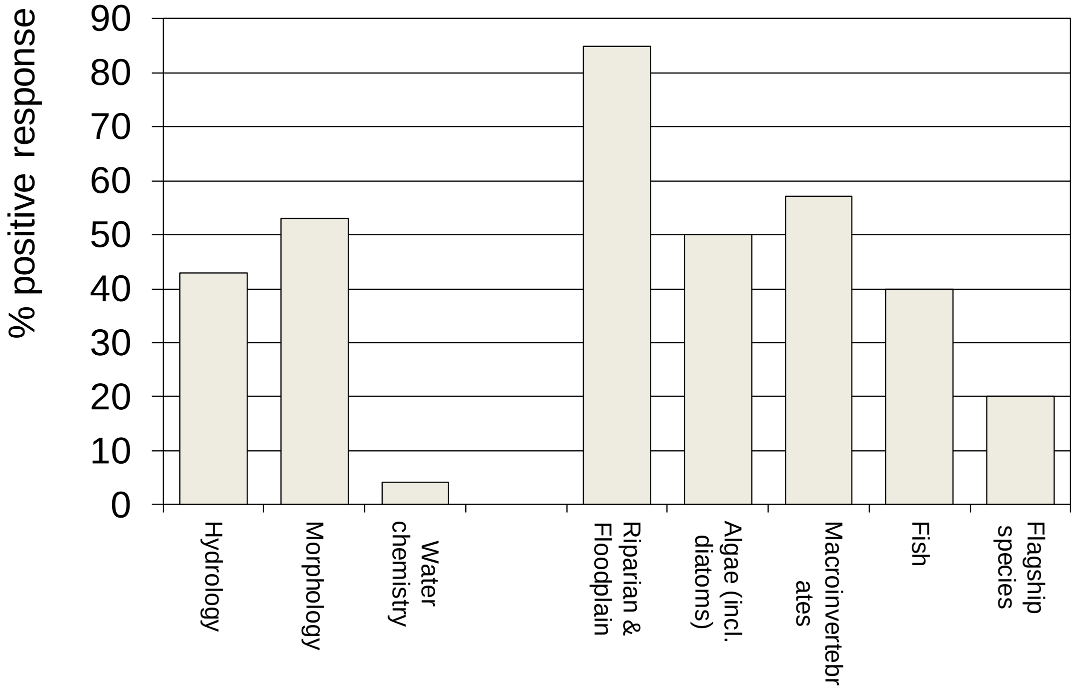


Figure 15. Percentage positive response reported in river restoration literature.

In lakes these numbers were higher (Figure 16). In total, 66% of the eutrophication restoration projects showed positive responses for phosphorus and/or nitrogen and related parameters. 64% of the biological organism groups included in the studies showed positive responses.
For marine waters data were not available.

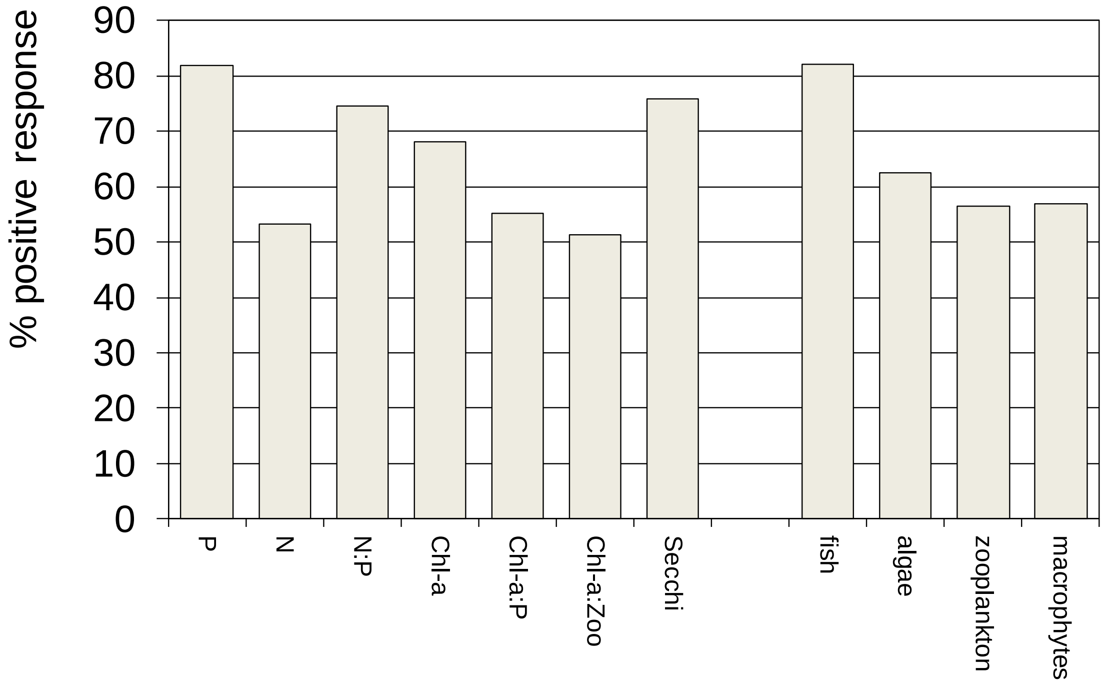


Figure 16. Percentage positive response reported in lake restoration literature.

Successful restoration is hard to define as end points and goals are often hardly described. Still, for each of the water categories studies are available that showed some indication of restoration success. But a number of studies also showed that ecological recovery takes time, can be delayed or even can fail. What restoration ecology needs is (see also Reitenberger et al. 2010):

- Definition of clear goals for restoration at catchment scale that are based on recent biological monitoring results and the actual distribution of targeted species or communities.
- Identification of best-practice restoration measures to address the specific pressures.
- Balancing all measures within a catchment in order to reach the best possible synergy effects of single component measures, and ultimately to achieve recovery of the entire catchment.
- Knowledge of indicators that can be monitored at large scale and be relevant for the measure taken.
- A monitoring design extracted from an experimental design that addresses the goals defined for restoration and that is likely to be successful at the large scale and in the long term.
- Pre-restoration monitoring as a basis for monitoring of progress, and ultimately of success.
- Indication of the time span for each measure to become successful.
- Monitoring of the post-restoration (abiotic) hydromorphological and biological developments based on before-after-control-impact surveys.

- Analysis of monitoring data according to state-of-the-art statistical techniques to identify potential shortcomings and to help to develop new indicators that also cover restoration effects on processes and community functions.
- Development of predictive models to support the design of future restoration projects and to assess their potential to become successful.

Over the next decades, annually, much money will be spent on restoration in order to improve and maintain the ecological status of rivers, lakes and estuarine and coastal waters. Whether these investments have the desired effect will depend on the quality of restoration measures taken and monitoring to adjust during the recovery process. Only a small fraction of the investment would be initially required to test the hypotheses defined on forehand and thereby, to establish a sound scientific and applicable basis for future restoration.

Chapter 6. Recovery: Organism groups

Rivers

Remeandering

Of the Biological Quality Elements (BQEs), macroinvertebrate (macrozoobenthos (MZB)) indicators were the most often applied during monitoring exercises (Figure 17). Following macroinvertebrates in descending order of frequency of use are fish, macrophytes and phytobenthos. Questions have been raised, however, about the reliability of macroinvertebrate indicators, particularly in the initial few years following river rehabilitation by various authors (Sporka et al., 2006, Blocksom & Flortmersch, 2008; Haase et al., 2008). In a study by Matthews et al. (2010), all types of rehabilitation intervention and the ability of different indicator groups to reveal progress towards restoration goals within five years were examined. The macro-invertebrate group were seen to perform relatively poorly compared to other indicators (Figure 18). Fish and macrophytes performed better but lagged behind other non-ecological indicators analysed. Of all biological indicators, terrestrial indicators monitored away from the river channel revealed early progress towards project goals the best.

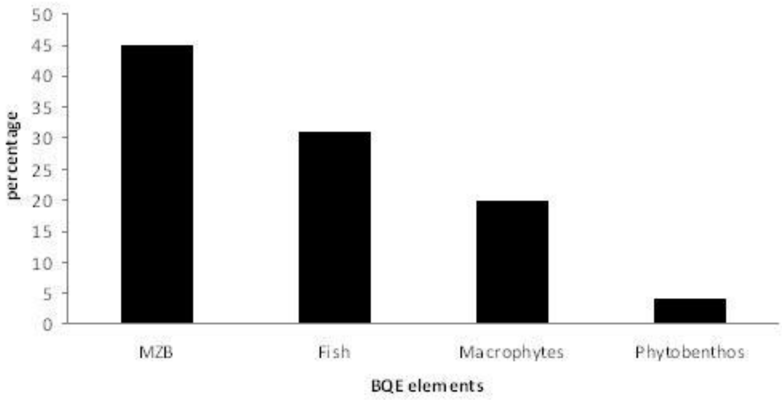


Figure 17. Percentage representation of BQE elements within monitoring schemes.

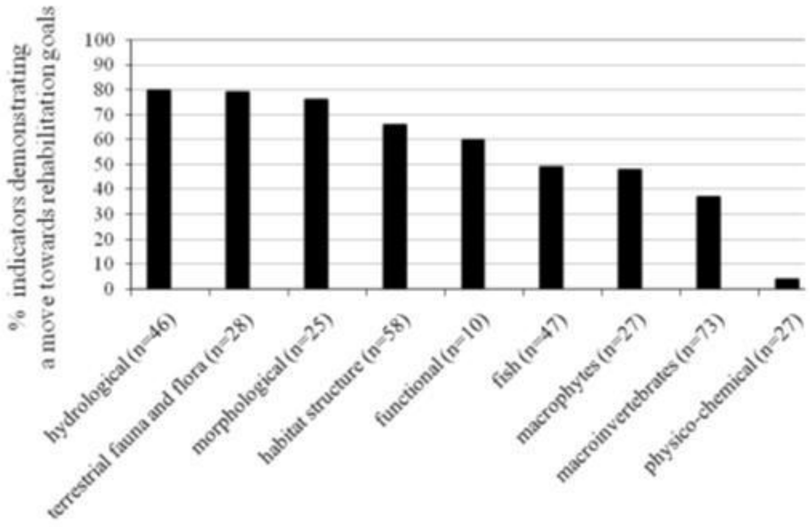


Figure 18. Positive indicator response per indicator group within the first five years of monitoring (Matthews et al, 2010).

Removal of weirs and dams

The biological impact of weir removal has been studied most often for benthic invertebrates (83% of all references), whereas aquatic macrophytes and fish were also frequently addressed (58 and 50%, respectively); phytobenthos has been rarely addressed (Figure 19).

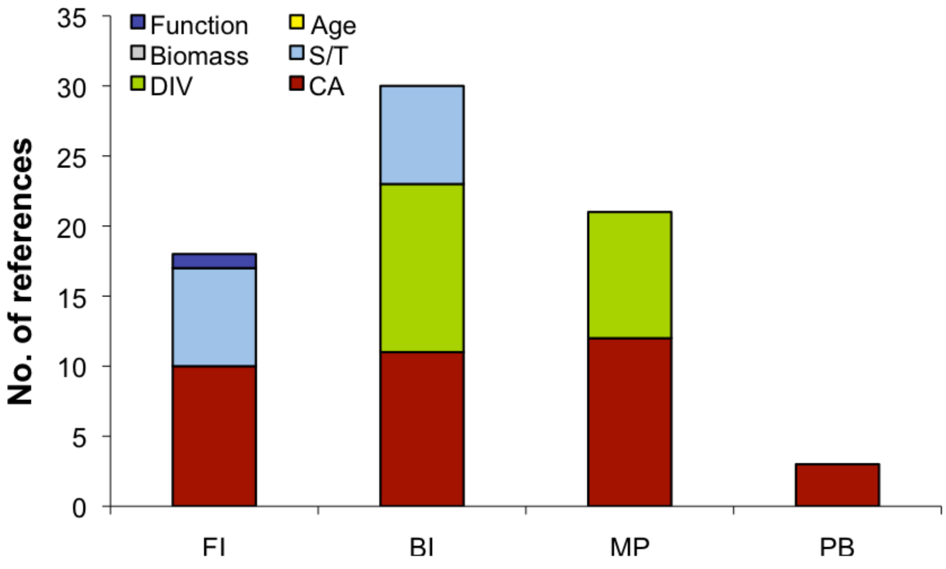


Figure 19. Number of references on weir removal addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB). (after Feld et al. 2011)

Riparian buffers

The majority of studies on riparian buffers adressed effects on benthic macroinvertebrates and fishes (Figure 20).

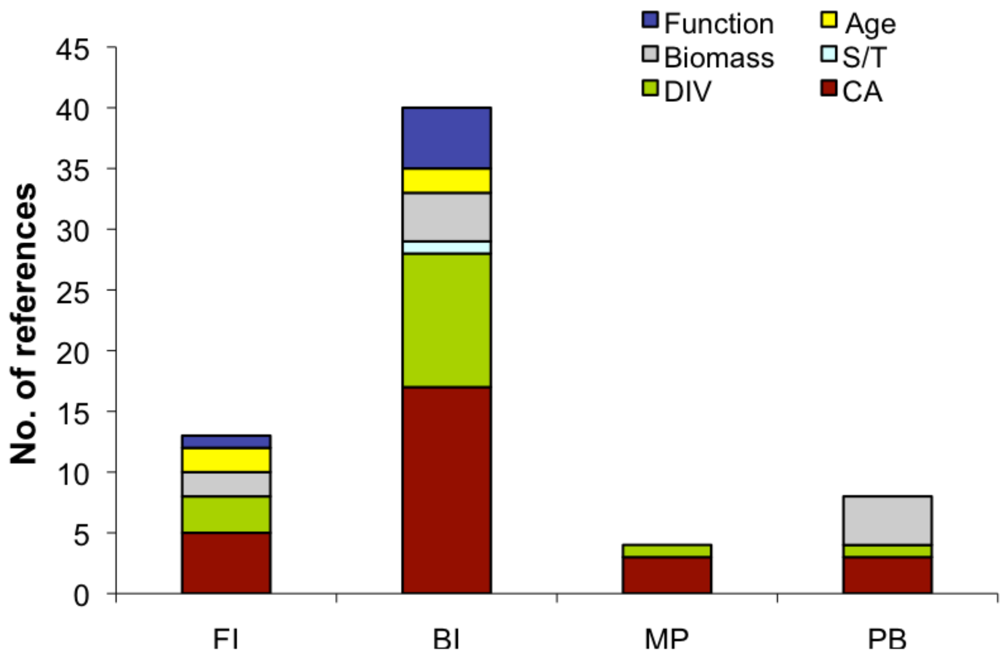


Figure 20. Number of references on riparian buffers addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB). As a study may refer to more than one community attribute, the overall number of references exceeds the number of 38 restoration references reviewed. (from Feld et al. 2011).

Enhancement of in-stream habitat stuctures

By far, most ecological effects on enhancement of in-stream habitat structures development were reported for the fish community attributes (21 references) followed by benthic macroinvertebrates (7), macrophytes and phytobenthos (2 each) (Figure 21).

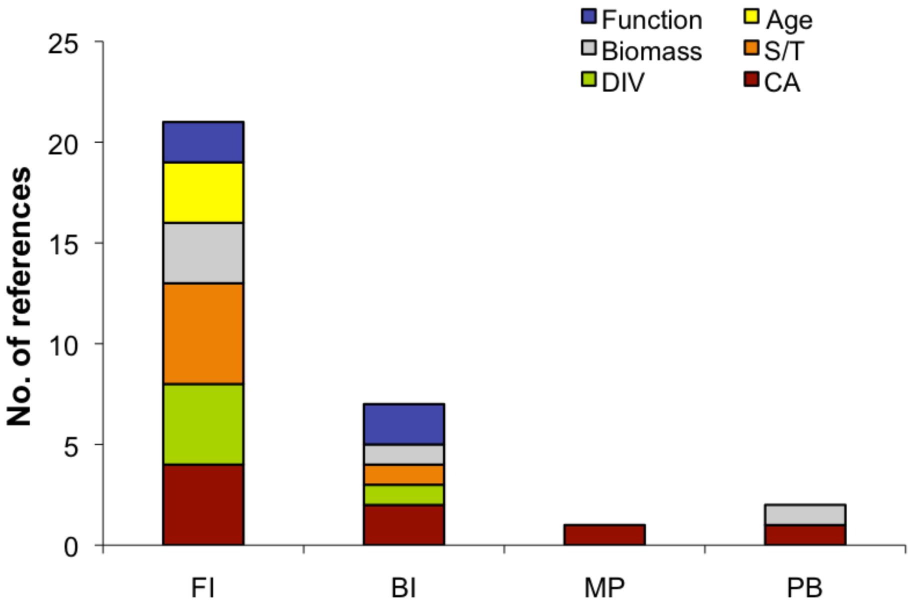


Figure 21. Number of references on enhancement of in-stream habitat structures improvement addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB). As a study may refer to more than one community attribute, the overall number of references exceeds the number of 75 restoration references reviewed.

Lakes

The literature review returned 333 lakes in which the recovery of at least one BQE was reported following external nutrient load reduction alone, 130 lakes in which only in-lake management was conducted and 51 lakes in which in-lake and external nutrient load management measures were conducted (Figure 22). Reports on phytoplankton were most common (44% of case studies reporting ecological recovery) followed by macrophytes (15%), zooplankton (14%), macroinvertebrates (13%), fish (12%), waterfowl (2%) and bacterioplankton (<1%).

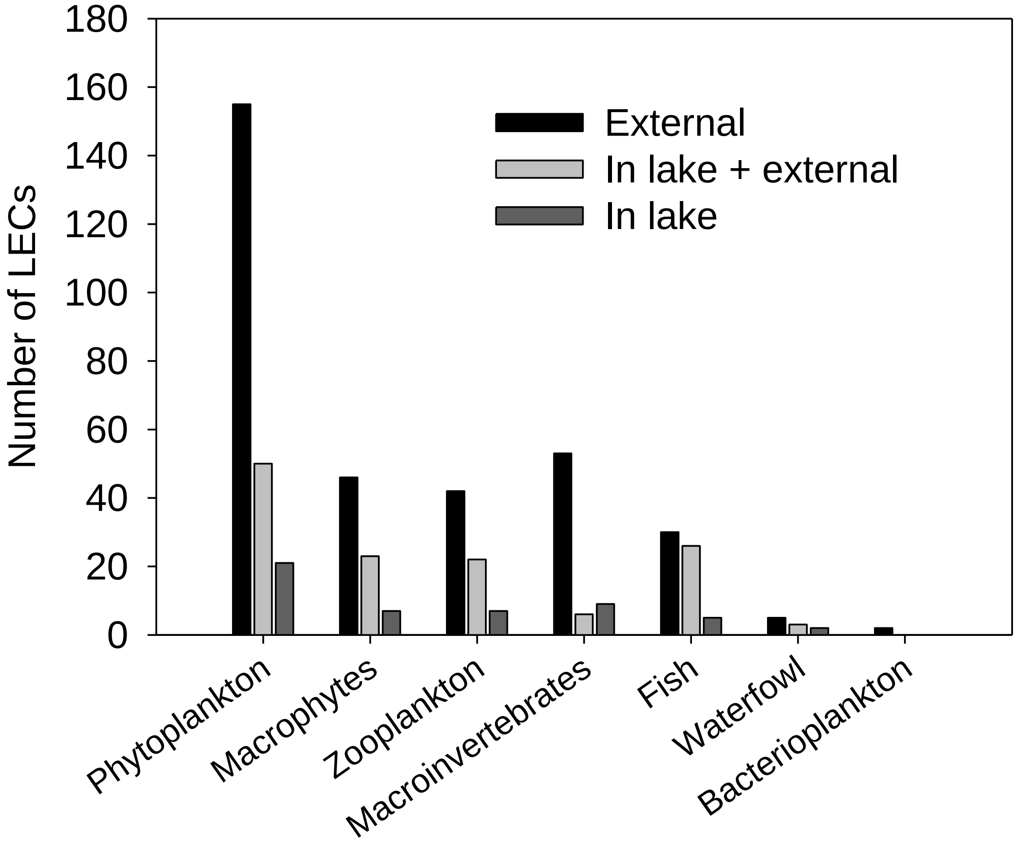


Figure 22. Number of lakes returned in the literature review for three management scenarios (external nutrient load reduction only, in lake management coupled with external nutrient load reduction, and in lake management only) for each of the 7 BQEs.

The study did not separate the individual restoration measures taken.

A number of studies have analyzed post-liming conditions of lakes and watercourses. As mentioned, chemical response is almost immediate, while response of different organism groups is taxon-specific. Fish, phytoplankton and benthic invertebrate assemblages are often monitored to determine the effects of liming on lake communities. Response times varying considerably and are often site-specific, but in general phytoplankton respond > benthic invertebrates > fish. Post-liming biological restoration has often focussed on two areas of study; namely, measures to facilitate natural recolonization and re-establishment of locally extinct populations and reintroduction of locally extinct species by restocking (Bergquist 1995). For example, removal of migration obstacles and improvement of habitat are two measures used to facilitate recolonization and establishment.

In contrast to assessing the effects of liming, fewer studies have looked at natural recovery of acidified lakes and watercourses. For lakes, fossil remains of diatoms and other organism groups (e.g. chironomid midges) have been frequently used in the Nordic countries, the UK and Canada. Although seemingly costly and requiring a substantial amount of taxonomic expertise, paleo approaches have been shown to be extremely good at establishing pre-acidified conditions as well as for tracking long-term changes in assemblage composition. More recently these approaches have been used to determine if recovery trajectories follow degradation pathways.

By contrast, use of contemporary data is someone limited by the scarcity of long-term monitoring data. Recent studies have shown that assessment of recovery is dependent on the response variable chosen, and that factors other than improved water quality can confound interpretation. Stendera and Johnson (2008) analyzing a dataset consisting of 10 boreal lakes and 16 years of continuous time-series data showed that several of the chemical and biological metrics showed positive trends over time, supporting the biological recovery. For example, phytoplankton diversity indicated signs of recovery. Findings from invertebrate assemblages were equivocal; littoral invertebrate assemblages showed positive trends, but similar trends were also evident in the circumneutral reference lakes, indicating that other factors than improved water quality might be driving these shifts in assemblage composition. In a similar study, Johnson and Angeler (2010) found that acidified lakes (n=4) had more pronounced shifts in assemblage composition than did reference lakes (n=4), indicating recovery over the 20 year time series. Similar to the findings of Stendera and Johnson (2008), the most marked differences were noted for phytoplankton assemblages. However, while trends in water chemistry showed unequivocal signs of recovery, responses of phytoplankton and invertebrate assemblages, measured as between-year shifts in assemblage composition, were correlated with interannual variability in climate (e.g. North Atlantic Oscillation, water temperature) in addition to decreased acidity. The finding that recovery pathways and trajectories of individual acidified lakes and the environmental drivers explaining these changes differed among assemblages shows that biological recovery is complex and the influence of climatic variability on recovery is poorly understood.

Estuarine and coastal waters

Borja et al. (2010) surveyed the current literature and identified 51 long-term cases where (1) actions were taken to remove or reduce human pressure effects; (2) information on the responses of biological elements was available; and (3) medium or long-term monitoring of the recovery occurred. In 38 out of these 51 cases benthic invertebrate were studied. Fish were studied in 8 out of 51 cases and macrophytes in 7 out of 51 cases. Macro-(algae) were studied in only two cases (Table 6).

Table 6. Overview of the number of references adressing the diffrent BQEs based on a review by Borja et al. (2010).

BQE	Number of studies
Benthic invertebrates	38
Fishes	8
(Macro)algae	2

Macrophytes/marsh	7
Birds	1

According to the literature review the most common BQE used to asses recovery are the benthic invertebrates which has been used to address different kind of pressures (see Table 3).

Summary

The majority of restoration studies in rivers and in estuarine and coastal ecosystems have focused on macroinvertebrates. In rivers also fish are important indicators. In lakes phytoplankton is the BQE studied most extensively. The difference in indicator groups used goes back to the causes of degradation. In lakes eutrophication is most important and phytoplankton best reflects the nutrient status of the lake over time. In rivers most degradation goes with hydromorphological change. Macroinvertebrates and fish respond strongly tot these types of changes. The choice of macroinvertebrates as indicators of degradation in estuarine and coastal waters is less obvious as eutrophication and saprobiphication are most common causes of degradation along with bottom disturbances. The latter would best be reflected in macroinvertebrate responses the first less. The confounding factor in estuarine and coastal waters for phytoplankton is water movement. Water movement reduces the indicative value of phytoplankton.

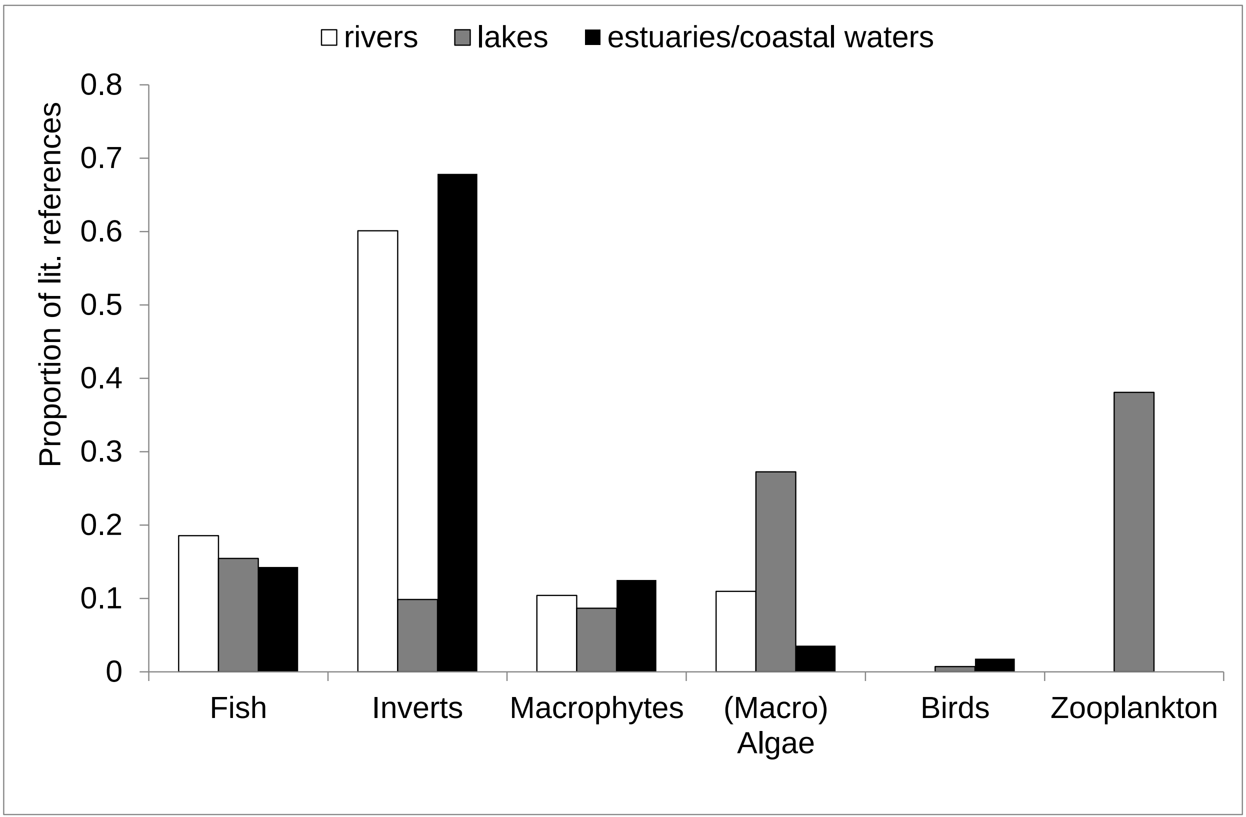


Figure 23. Proportion of literature references that reported on specific biological quality elements in river, lake and estuarine and coastal water restoration studies.

Chapter 7. Recovery: Time-scale

Rivers

A significant confounding factor when considering the assessment of successful recovery is the length of the post-management monitoring period in relation to the reported transient recovery period for all indicator variables across all management approaches.

The majority of all 168 papers on river restoration studied by Feld et al. (2011) only deal with short-term effects of restoration (<5 years). Per measure the following recovery or study periods were noted:

Remeandering

Time-scales for recovery after remeandering could not be derived from the meta-analysis of existing literature performed by Feld et al. (2011), due to limited data availablity. Although limited data constrained the analyses, it was concluded that non-ecological indicator groups (hydrology, morphology) react quickly giving a positive response to intervention within two years follwing restoration. 90% of the non-ecological indicators responed positively to restoration within the initial two years follwowing restoration. No trend indicating a particular time period where ecological indicators demonstred a response could be observed.

Removal of weirs and dams

The findings presented here are consistent with the conclusions of Doyle et al. (2005): each variable develops in a specific time scale after weir (dam) removal. The re-establishment of the longitudinal connectivity that allows migratory fish to move is almost immediate. Many environmental parameters such as substrate conditions and the overall water quality may recuperate within a few years, while water temperature will change almost instantly. In contrast, biological recovery in general requires several years or even decades after removal and is expected to occur once the fine sediments have been transported farther downstream (e.g., Thomson et al. 2005). This effect strongly depends on the quantity of sediments that were accumulated upstream of the barrier, on the water velocity, on the gradient of the riverbed and eventually on the specific technique of weir (dam) removal (Bednarek 2001). According to the author, full recovery may take up to 80 years, but the literature does rarely include monitoring periods longer than five years after implementation of a measure. The time-scale of recovery after weir removal continues to remain speculative unless long-term monitoring is being conducted to provide evidence.

Riparian buffers

Time-scales for recovery after ripirian buffer re-establishment could not be derived from the meta-analysis of existing literature performed by Feld et al. (2011), because two thirds of the studies were conducted less than 10 years after the measures were taken. Studies over longer time spans are almost completely lacking (Figure x). This time-scale may be sufficient to detect the direct effects of in-stream habitat improvements, but it is likely to be insufficient to detect major indirect effects of riparian buffer restoration on the overall functioning, on important processes such as wood recruitment and the supply of energy to the in-stream food web. But some theoretical considerations imply the timescales required for riparian buffer management to achieve maturity and to provide all relevant ecological functions. Riparian trees like black alder and willow require 30–40 years to mature and eventually reach their final height and maximum canopy cover (Jowett et al. 2009). This time frame is probably required to provide nutrient and sediment retention, and temperature control. Longer time spans are needed to start providing natural amounts of LWD and, hence, to gain the desired effect on the in-stream hydromorphology (bed form processes and habitat improvement).

Instream mesohabitat enhancement

Feld et al. (2011) reported that the time span between restoration and monitoring of effects was highly variable and ranged from 1 to 50 years (this included both biotic and abiotic response). Feld et al. (2011) also concluded that the majority of restoration studies on in-stream mesohabitat improvements are short-term and, thus, cannot provide insight in long-term effects. Also, the few references including long-term monitoring results largely imply that habitat enhancement and related biological effects of the most frequent restoration measures are prone to environmental impacts beyond the scale of restoration. Irrespective of the many (short-term) positive effects, in particular due to the introduction of large woody debris (LWD), roughly half of the reviewed references imply failures of habitat improvement, biological recovery or both, when it comes to long-term recovery. This does not mean these studies reported failures, but we simply cannot judge the level of success of many studies because of their limited monitoring efforts (Feld et al. 2011).

Lakes

Eutrophication reduction

The 46 lake equivalent case studies showed that only 45% of the 31 lakes reported post-management monitoring periods of 1 year or less in excess of the reported transient recovery periods. Therefore, it is questionable whether or not true recovery had been reported in these studies (Figure 24).

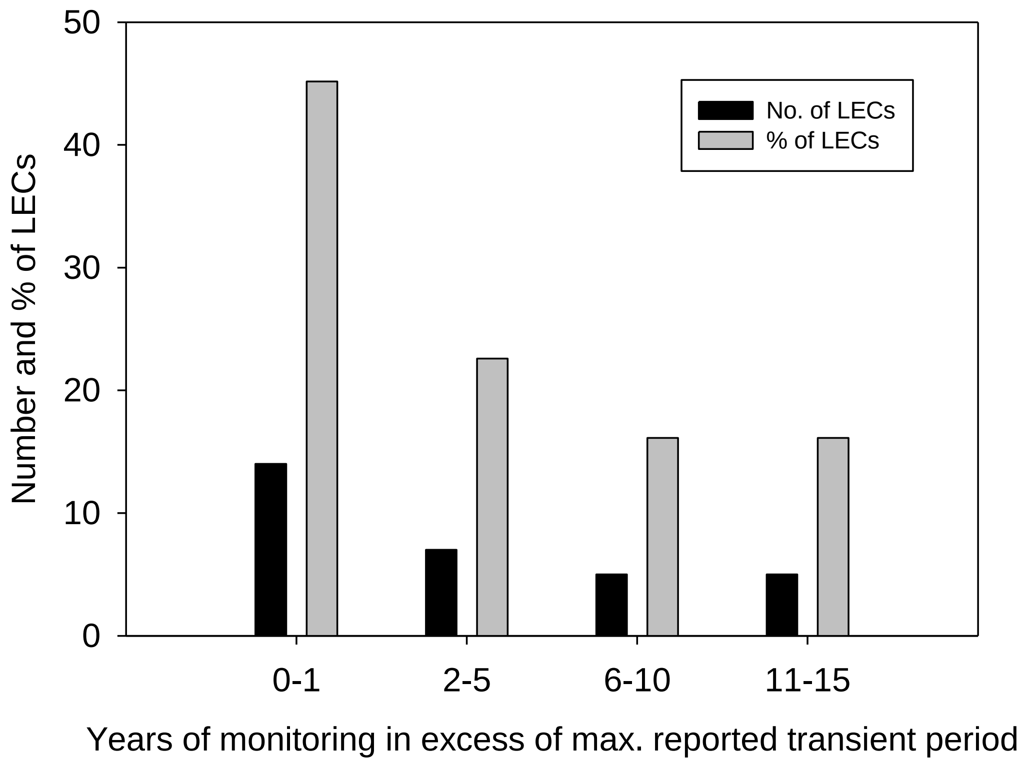


Figure 24. Distribution of lake equivalent case studies (n = 36) for which monitoring was conducted in excess of the estimated maximum transient recovery period.

More in detail, the responses of the bacterioplankton community following eutrophication management (Table 7) were summarised from only 3 publications. The responses of the macroinvertebrate community following a review of 68 publications, the responses of the macrophyte, fish and zooplankton community following a review of 76, 61 and 71 publications respectively.

Table 7. Summary of the findings of 364 peer reviewed publications in terms of the timescales and trajectories of structural and functional ecological recovery of lakes after eutrophication management. References were retrieved from Web of Science using the search term [Lake+Eutrophication+Recovery] in January 2011.

Organism group	Main changes in environmental state	Impacts on ecological structure	Impacts on ecological function	Factors confounding recovery
Bacterioplankton (<18 years)	Decrease in TP leading to change in zooplankton community	Unclear – potential increase due to DO increase but potential decrease due to higher grazing rates by *Daphnia*	Increase or decrease in contribution to energy transfer	Unknown
	Increase in DO			
	Decrease in labile DOC through reduction in planktonic production			
Phytoplankton (2-20 years)	Reduction in TP concentration	Overall reduction in phytoplankton biomass (especially in spring and summer). Reduction may be slower in summer due to internal loading	Decrease in quantity and increase in quality of food for zooplankton	Sustained P availability in metalimnion of stratified lakes leading to dominance of metalimnetic cyanobacteria
	Increase/decrease in TN concentration	Reduction in spring diatom biomass	Decreased organic matter deposition to the sediment	N limitation prior to restoration leading to no response in phytoplankton community
	Increase/decrease in Si concentration	Increase in importance of diatoms, cryptophytes and chrysophytes in shallow lakes	Lower mass of nutrients available for internal nutrient cycling	Increase in Si concentrations in summer caused by P limitation of diatoms in spring
		Reduction in relative contribution of cyanobacteria to total phytoplankton community at meso-oligotrophic end point TP concentrations	Increase in DO concentrations at the sediment surface	Increased external loading and persistent internal loading
		Replacement of non-heterocystous cyanobacteria with heterocystous cyanobacteria following strengthened N limitation at mesotrophic end point TP concentrations	Increased resilience to negative switches from clear water conditions	Natural and human induced changes in fish stock leading to trophic cascades through zooplankton

		Decrease in cyanobacteria and an increase in relative contribution of dinophytes and chrysophytes in deep lakes	Lower risk of cyanobacterial toxin production	Feedback mechanisms between buoyancy regulating cyanobacteria and sediment P cycling in poorly flushed lakes
		Species richness increase towards mesotrophic TP end point	Higher overall genetic diversity	
Zooplankton (1-17+years)	Reduction of TP concentration	Increase in *Daphnia* and large cladoceran relative abundances (e.g. *Daphnia hyalina* and *D. galeata*) in relation to smaller taxa (e.g. *D. galeata, D. ambigua, Ceriodaphnia pulchella*), and especially in relation to rotifer biomass, as nutrient conditions change from hypertrophic to mesotrophic	Increased relative grazing pressure on smaller "edible" phytoplankton and bacterioplankton	Presence of predatory zooplankton as a result of top down effects of fish
	Increased macrophyte cover as *refugia*	Increase in zooplanton:phytoplankton ratio	Higher intensity clear-water phase in spring	Fish stocking with zoplanktivorous fish
	Increase in edible phytoplankton relative to cyanobacteria	Increase in cladoceran species richness	Increased resilience of ecosystem to reverse switch through control of phytoplankton	Recovery of zooplanktivorous fish following biomanipulation
	Decreased zooplanktivorous fish abundance relative to piscivorous fish	Increase in cladoceran body size	Greater relative importance as a nutrient source at low nutrient concentrations	Inputs of pesticides
		Decrease in total zooplankton biomass.	Higher overall genetic diversity.	Inputs of other industrial pollution.
		Increase in oligotrophic indicator species including *Ceriodaphnia cornuta*, *Daphnia gessneri*, other small cladocerans like *Moina micrura*, *Bosminopsis deitersi*, *Notodiaptomus cearensis* and oligotrophic calanoid copepod species towards oligotrophic conditions	Decreased organic matter deposition to sediments	Occurrence of marine macroinvertebrate grazers following salinisation

Macroinvertebrates (10-20 years)	Reduction in organic matter to sediment	Reduced overall abundance.		High benthivorous fish abundance
	Increased DO concentrations in benthos	Increased species richness and diversity		Stratification leading to anoxia in benthos
	Change in grazing pressure related to fish community change	Plecoptera, Ephemeroptera, Coleopteran, Trichoptera all increased in relative abundance		Increase in external loading and/or persistent internal loading
	Expansion of macrophytes into deeper waters	Increase in relative abundance of indicator taxa, e.g. Cladocera, gastropods & Hydracarina - "oligotrophication". More extensive colonisation of deeper water linked to macrophyte recovery.		Persistent organic sediment loading
		Increase in chironomid to oligocheate ratio		Invasion by dreissenid mussels
Macrophytes (2-40+ years)	Reduction in TP and TN concentrations	General shift from macroalgae (e.g. *Cladophora* and *Enteromorpha* spp.) => tall angiosperms (e.g. *Potamogeton pectinatus*; *Myriophyllum spicatum* etc.) => short angiosperms (e.g. *Eleocharis acicularis and Littorella uniflora*), characean macrophytes (e.g. *Chara globularis* and *Nitellopsis obtuse*) and mosses (e.g. *Fontinalis antipyretica)* as nutrient concentrations are reduced from hypertrophic to oligotrophic conditions	Decrease in sediment P release at low-moderate biomass leading to aeration of sediments or an increase in sediment P release at high biomass; this results in hypoxia in benthos or reduction in sediment disturbance leading to decrease sediment P release	Increase in external loading and/or persistent internal loading
	Reduction in phytoplankton biomass	Increase species richness towards mesotrophic conditions.	Partitioning of P from phytoplankton to macrophyte biomass	Grazing by herbivorous waterfowl (e.g. coot) and fish (e.g. bream and roach)
	Increase in water clarity	Increase macrophyte colonisation depth towards meso-	Increase in benthic primary production	Habitat disturbance due to wave action

		oligotrophic conditions		and water level fluctuations
	Improvement in substrate quality	Increased frequency of occurrence towards mesotrophic conditions	Increase in *refugia* for benthic and planktonic organisms.	Macrophyte control by humans using mechanical harvesting or herbicides.
	Reduction in herbivorous waterfowl and fish	Macrophyte abundance decreases above about 2 mg N l^{-1} due to competition for light with epiphytes	Decrease in water column NO_3-N concentrations through direct uptake and enhanced denitrification	Invasive species ingress or extinction of regional seed bank or blocked distribution pathways
	Reduction in periphyton shading		Increase in food supply for waterfowl and fish	
Fish (2-10+ years)	Decrease in TP concentrations	Shift from cyprinids to percids to coregonids to salmonids with decreasing TP	Competitive advantage for visual predators	Biomanipulation and/or invasive non-native species
	Increase in water clarity	Fish species richness increases towards 0.1 to 0.4 mg TP l^{-1}	Increased predation pressure on zooplanktovorous fish	Persistent internal loading
	Increase in macrophyte abundance	Increase in littoral fish species (e.g. gudgeon, rudd, and pike) relative to pelagic species (e.g. pikeperch and ruffe)	Increased energy transfer through littoral habitats	Climate change related temperature increases, especially in winter and spring
	Decrease in zooplankton biomass	Decrease in fish abundance with decreasing TP	Increase in chlorophyll:phosphorus ration as a result of trophic cascade	Blocked distribution pathways
Waterfowl (2-21+ years)	Reduction in TP concentrations	Increase in herbivorous bird species including coot, goldeneye, and pochard	Increased energy transfer to waterfowl	Competition for food with bream
	Reduction in benthivorous fish species	Increased benthivorous birds	Increased grazing on macrophytes	Persistence of internal loading delaying recovery of macrophytes and macroinvertabrates
	Increase in macrophyte abundance		Increased nutrient inputs to lakes	Extreme fluctuations in water level
	Increased abundance of macroinvertebrates		Increased nutrient cycling from macrophytes through waterfowl to water column	

The time it takes for a lake to recover after reduction of nutrient loading varies considerable between organism groups, but also within organism groups (Table 7). Phytoplankton responded to reduced nutrient loading within two to 20 years. Similar recovery times were recorded for zooplankton (1-17+ years) and waterfowl (2-21+ years). Recovery of the macroinvertebrate community takes longer and varies between 10 and 20-years. Macrophyte recovery varied between 2-40 years and fish recovery between 2 and 10 years. These numbers are based on the analyses performed by Spears et al. (2011). The respective recovery times indicate that a rapid recovery is possible for most organism groups, except for macroinvertebrates, but in many cases recovery takes longer. The differences in recovery time between lakes for a single organism group can have several causes. One major cause is the diffrences in internal P loading between lakes. In shallow lakes, in-lake biogeochemical processes (Figure 11) can regulate reductions in TP concentrations leading to changes at the seasonal, annual and decadal scales. Jeppesen et al. (2005b) reviewed the recovery of 35 lakes following external nutrient load reduction and estimated that internal P loading delayed the recovery of lakes between about 10-15 years. However, examples are also available in the literature of lakes in which internal loading has delayed recovery for up to 20 years (e.g. Lake Søbygård, Denmark; Søndergaard, 2007) post external load reduction. Only submerged macrophytes seem to respond slowly to lake restoration, which is in accordance with many other findings (Strand 1999; Jeppesen *et al*. 2005; Hilt *et al*. 2006), although a fast response of the plants has occurred in other case studies (Hansson *et al*. 1998, Søndergaard et al. 2007). In general, the timing of the transient period is known to be driven by a range of factors including retention time, pollution history, sediment P composition and concentrations, and depth (Sas, 1989).

Biomanipulation

In more than half of the 70 biomanipulation projects studied by Søndergaard et al. (2007), secchi depth increased and chlorophyll-a decreased to less than 50% within the first few years. In some of the shallow lakes, total phosphorus and total nitrogen levels decreased considerably, indicating an increased retention or loss by denitrification. The strongest effects seemed to be obtained 4–6 years after the start of fish removal. Søndergaard et al. (2007) state that the long-term effect of restoration initiatives can only be described for a few lakes, but data from biomanipulated lakes indicate a return to a turbid state within 10 years or less in most cases. One of reasons for the lack of long-term effects may be internal phosphorus loading from a mobile pool accumulated in the sediment.

Recovery from acidification can be rapid with lake liming, but substantially longer when catchment or wetlands are limed. Short-term post-liming effects are characterized by rapid expansion of individual populations, attributed to low competition and predation and a surplus of resources (e.g. nutrients). For example, changes in light regimes may result in increases in phytoplankton biomass 1-2 months after liming, with even blooms occuring (e.g. Svensson et al. 1995 chapter 10), rapid development of macrophytes such as *Myriophyllum alterniflorum*, whereas other species may take decades to recolonize (Larsson chapter 7) and rapid expansion of certain fish populations (Degerman et al. 1992). Long-term effects have, however, been largely attributed to biotic interactions such as competition and predation. However, as discussed above, liming needs to repeated periodically and if lapsed and buffering capacity falls a single acidic episodes can eradicate years of recovery (Ormerod and Durance 1992). As discussed by Angeler and Goedkoop (2010), repeated lake liming events may also be seen as a form of pulse disturbance, resulting in less complex food webs compared to circumneutral lakes.

Time lags associated with natural recovery from decreased deposition of acidifying compounds are much longer compared to lake liming. Although many lakes are showing improved water quality, episodic acid events continue to occur in poorly buffered areas, since soils are still affected by acidification. In contrast to regional improvement in water chemistry, studies documenting changes in biota are scarce and findings equivocal. Moreover, although not tested, response times appear to be dependent more on site-specific and regional factors such as changes in wet-dry years driven by the North Atlantic Oscillation. In numerous cases the data sets used to monitor ecological recovery are not of sufficient length, with a number only now becoming long enough to assess the biological response. It is unclear to what extent the patchy recovery observed reflects the availability of high quality records as opposed to real limits to the recolonisation and re-establishment of sensitive organisms. A continuation of existing monitoring programmes is essential together with a focus on how communities are responding structurally and functionally to improved water chemistry and the effects that other confounding factors may have on this.

Estuarine and coastal waters

Borja et al. (2010) reviewed 51 studies (Table 8). They concluded that meiofauna may need only several months to recover, whereas hard bottom macroalgae and some seagrass species can take more than 22 years. Birds may take even more time, until 70 years. Fish assemblages appear to recover from most pressures in less than 10 years, although it may take several decades to acquire a full species complement after starting from a state without any fish community. In all cases the time to recovery will depend on the type of restoration.

Table 8. Time span of recovery per BQE after restoration or removing of pressure based on a review of 51 studies by Borja at al. (2010).

BQE	Recovery time
Benthic invertebrates	From months to 20 years
Fishes	1- 20 years
Macroalgae	14 - >22 years
Macrophytes/marsh	2-20 years
Birds	15-70 years

When recovery times are related to pressures (Table 9) it seems severe impacts, whether acute, such as large oil-spills, chronic (low level inputs) or persistent over time and space (such as sewage sludge disposal, extensive wastewater discharge or mine tailings), require periods up to 10-25 years for complete recovery. Conversely, restoration after physical disturbance (including dredging and restoration of tidal inundation) that does not leave a “legacy” stressor such as a persistent contaminant can take 1.5 years for recovery, although some sensitive organisms (such as angiosperms) may take over 20 years to recover (Table 9).

Table 9. Time span of recovery after restoration or removing of pressure based on a review of 51 studies by Borja at al. (2010).

Source of stress	**Recovery time (years) after removal of stressor (*)**
Physical - marsh restoration/land claim reversal	<5
Physical - navigation dredging, aggregate extraction and material disposal	<5
Organic enrichment - sewage, oilspills, paper waste	5-15
Chemical pollution - sediment contamination	>15

(*) depending on the BQE

Borja et al. (2010) mention that the studies included in their review focus on an initial reappearance of particular biological elements. Borja et al. (2010) noted that the presence of a biological element following colonisation is not necessarily an indication that a fully functioning ecosystem has been created (Mander et al. 2007, Mazik et al. 2007). Although the analysis shows that in some cases recovery can take <5 years, the full recovery of many coastal marine and estuarine ecosystems can take a minimum of 15–25 years from over a century of degradation and attainment of the original biotic composition and diversity and complete functioning may lag far beyond that, possibly at least another 25 years. Some ecosystems may never attain the technical definition of being restored, but end up irreversibly in an alternative state, as shown in the Nervión estuary (Borja et al. (2010).

The time-span of recovery after removal of the pressure is highly variable and depends on the pressures and BQEs. Although in some cases recovery can take <5 years, especially for the short-lived and high-turnover biological components, full recovery of coastal marine and estuarine ecosystems from over a century of degradation can take a minimum of 15–25 years for attainment of the original biotic composition, diversity and complete functioning may lag far beyond that period.

Summary

Studies dealing with long-term recovery in rivers, lakes and marine ecosystems are scarce. Despite this gap in data recovery times were evaluated (Figure 25). One important question that also has to be answered before time spans of recovery can be compared between water categories is `How is recovery defined?´ In the review by Feld et al. (2011) recovery was in most cases based on qualitative linkages, indicating the direction of a trend (e.g. an increase of x causes y to decrease). For lakes recovery also seems to be an increase or decrease of aspects of the biological community as a result of changes in environmental (state) variables due to restoration. Johnson and Angeler (2010) defined recovery of lakes from acidification as when the interannual variability of the acidified site falls within the uncertainty of undisturbed reference sites. Borja et al. (2010) mention that the studies in estuarine and coastal waters included in their review focus on an initial reappearance of particular biological elements. This means none of the reviews actually addressed 'full recovery', including functional characteristics, inter- and intra-specific interactions such as predator–prey relationships and competition or the historical state or even an alternative good state. Remarkable, is that both marine and riverine literature address the issues related to the definition of recovery, while in lake literature this discussion seems to lack and there is a general consensus on shift from turbid to clear water as major goal.

The proportion of references indicates that most monitoring did not yet last longer than about 5 to 10 years. Only few studies (one each) in rivers and marine waters extended 20 years.

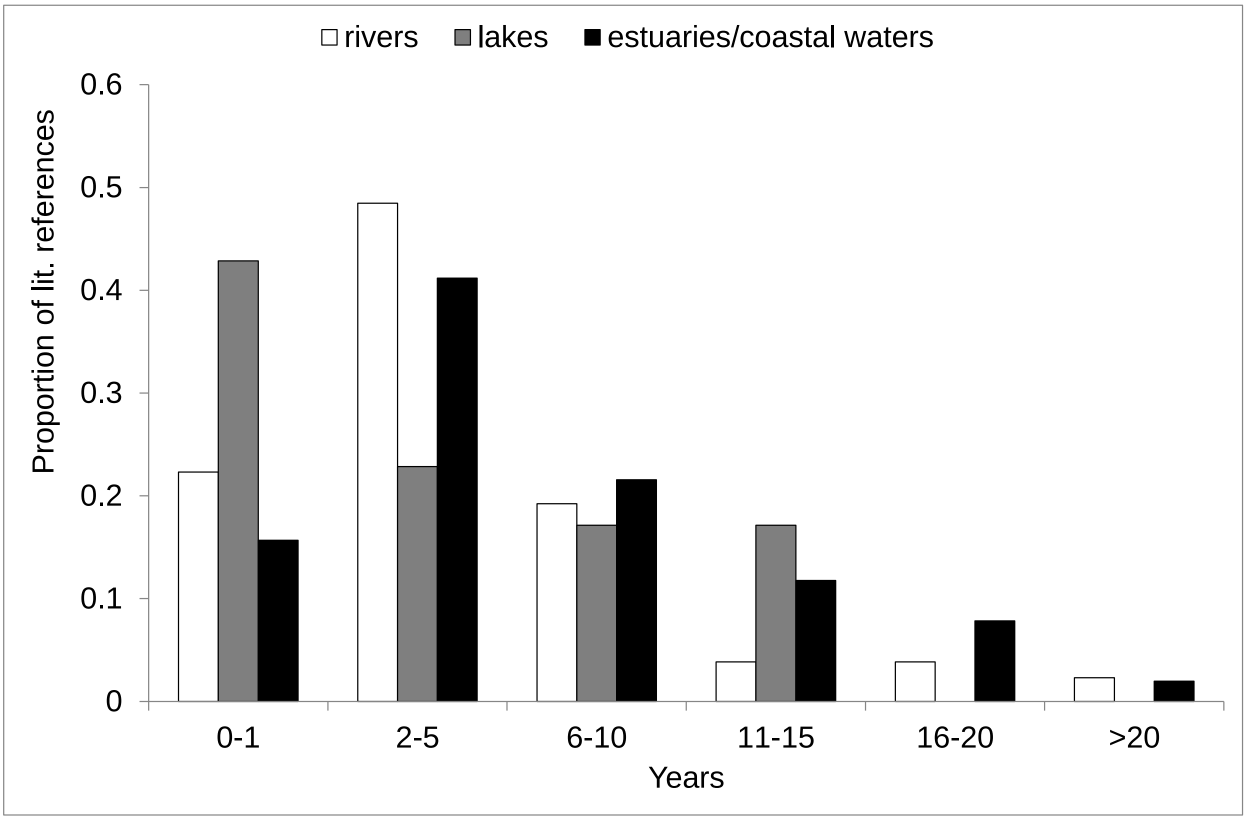


Figure 25. Proportion of literature references that indicated the time period of monitoring recovery in river, lake and estuarine and coastal water restoration studies.

Although, analyses in the different reviews do not address full recovery', authors do give indications on 'full recovery' based on estimates (Table 10).

Table 10. Estimates on the time required ro reach 'full recovery' of rivers (Feld et al. 2011), lakes (Spears et al. 2011) and estuarine and coastal waters (Borja et al. 2010) after restoration.

	Rivers	Lakes	Marine
Bacterioplankton		<18	
Phytoplankton		2-20	
Macroalgae			14->22
Zooplankton		1-17+	
Meiofauna			Months
Macroinvertebrates		10-20	months-20
Macrophytes		2-40+	2-20
Riparian veg.	30-40		
Fish		2-10+	1-20
Birds		2-21+	15-70

Marine ecosystems may take between 35 and 50 years to recover (Borja et al. 2010). Bednarek (2001) suggest recovery after weir removal may take as long as 80 years. Recovery after riparian buffer instalment may take at least 30-40 years.
Despite the fact that they do not indicate ‚'full recovery' we compared recovery times between the three water categories as mentioned in the different reviews. In marine ecosystems benthic invetrebrates and macrophytes have the potential to recover within months (in two studies on recovery of sediment disposal) and fish within one year. When only marine studies that recover from eutrophication are included, recovery times for macroinvertebrates varied between >3 years and >6 years. Although in some cases recovery can take <5 years, especially for the short-lived and high-turnover biological components, full recovery of estuarine and coastal ecosystems from over a century of degradation can take a minimum of 15–25 years for attainment of the original biotic composition, diversity and complete functioning may lag far beyond that period. In lakes recovery time from eutrophication for macoinvertebrates varied between 10 and 20 years. As in marine ecosostems recovery of macrophytes (2 to >40 years) and fish in lakes (2 to >10 years) be relatively fast. Response times for organism groups in rivers are lacking, because the literature rarely includes post hoc monitoring of more than 5 years. Also, the fact if biological response in rivers occurs within short term is undecided. Roni et al. (2008) stated that the potential benefits of most in-stream structures will be short-lived (<10 years) unless coupled with riparian planting or other process-based restoration activities supporting long-term recovery of key ecological and physical processes (Feld et al. 2011).

In both rivers and lakes the success rate of restoration measures appears to be much higher for the abiotic conditions than for the biotic indicators. Since eutrophication is considered to be the most important pressure in rivers and lakes, only this is not addressed in rivers, this might be a major cause. Especially, the response of macroinvertebrates in rivers is questionable, some studies mention recovery times of others question recovery of macroinvertebrates completely. In lakes internal nutrient loading often delays recovery.

A different case is biomanipulation. Based on the results of Søndergaard et al. (2007) restoration of lakes through biomanipulation (in particular fish removal) should be regarded as maintenance rather than as restoration. Fish removal in shallow eutrophic lakes, has had marked short-term effects on lake water quality, secchi-depth and chlorophyll-a in many lakes in the Netherlands and Denmark. However, Søndergaard et al. (2007) state that long-term effects (>8–10 years) are less obvious and a return to turbid conditions is often seen unless fish removal is repeated. As such we confirm that fish removal should be seen as a form of maintenance instead of restoration, because it has to be repeated every few years to maintain the effects previously acquired. The same conclusions were drawn on restoration of acidified waters. When the acidification sources are not tackled the measures need to be repeated.

The time-span of recovery after removal of the pressure/stressor is highly variable in all three water categories and depends on the pressures/stressors, especially if some are still present, and on the organim group(s) taken into account. Even the time span of recovery can range enourmously for a single organism group. The ranges in recovery times can be attributed to several factors delaying or confounding recovery. Especially, different water types exposed to different combinations of stressors respond differently to recovery. Furthermore, there needs to be agreement upon the restoration goals for the system and also what criteria will be used to determine attainment of the desired or targeted system (Simenstad et al. 2006). For example, it must be known whether a system is restored merely for its abiotic features, its structural elements, i.e. the appropriate species, or if full functioning.

Chapter 8. Recovery: Failure or delay in response

Rivers

Re-meandering

The first important reason for a failure or delay in re-meandering success was often that only one intervention incorporated within a set of interventions applied to a project stretch. Therefore, it was difficult to establish a direct link between re-meandering and observed changes. This is further complicated by a lack of the use of a control in many monitoring programs and a lack of standardization within monitoring. Other limiting factors such as floods, droughts, dispersal will mean that population; community and ecosystem responses to the addition of habitat will often take considerable lengths of time (Lake, 2001). These sorts of delays do not cause restoration to fail, but instead, may push response times beyond those over which monitoring is typically funded (Bond & Lake, 2003). External influences that were identified to have influenced the ecological recovery of the rivers studied were the presence of upstream source populations for colonisation, upstream management practices, water quality with particular emphasis on nutrient enrichment, large scale hydrological change and associated effects on sedimentation and erosion and project size. In general, factors such as water quality, source populations and hydrological characteristics are examples of variables that influence the predictability of river system recovery (Jähnig et al. 2010; Walsh et al. 2005).

Removal of weirs and dams

Many organisms are limited in their recovery by restricted habitat availability and potential limitations to reach the restored habitats (e.g., barrier structures), both of which are considered to be key limitations for recovery. A re-establishment of habitat variability requires geomorphological processes similar to pre-damming conditions (Doyle et al. 2005). Such natural geomorphological processes are required to enable fish reproduction, which is often limited or even inhibited due to the absence of suitable habitats to complete their life cycle (i.e. habitat for spawning, nursery, foraging). If geomorphological degradation, however, is irreversible, ecological recovery will hardly be possible without controlling quasi-natural geomorphological and hydrological processes. Even negative effects of weir and dam removal have been reported to last at least up to 5 years, or even more (Bednarek, 2001). Another limiting aspect refers to the size of a weir or dam. Orr et al. (2006) concluded that the effect of the removal of small dams was rather small compared to the natural variability of the entire system (Boulder Creek, USA). This finding suggests that small weir removal measures are not likely to have long-term deleterious effects (see also Thomson et al. 2005) and is as such not a limiting factor.

In-stream mesohabitat enhancement

Several examples of non-effect studies ended their discussion with the assumption that the absence of biological recovery was owed to continuing pressures at larger scales that were not mitigated by restoration, such as water quality problems (e.g., Pretty et al. 2003) or fine sediment entries due to intensive land use upstream (e.g., Larson et al. 2001; Levell and Chang 2008) that 'spoiled' and limited restoration effects. As Feld et al. (2011) stated "local habitat enhancement measures are often swamped by reach- or watershed scale pressures upstream that continue to affect the treated sites". The same applies for hydrological and geomorphological processes, which are often neglected (e.g. Beechie et al. 2010), but which constitute important characteristics that determine the level of success of restoration (peak discharges due to a high degree of imperious areas upstream).

River recovery failure or delay

The lack of biological recovery following river restoration measures has been attributed to ecological constraints that may have limited or even inhibited recovery in restoration studies (e.g. Jähnig et al. 2009a, Lorenz et al. 2009). Such constraints probably include meta population dynamics of available source populations (e.g. Shields et al. 1995b, 2006) and the dispersal capabilities of the community members to recover (e.g. Shield et al. 1995), although evidence of the underlying mechanisms is still lacking from restoration studies. The presence of source populations in the catchment and the absence of barriers blocking migration pathways are crucial for the colonisation of recovering species. Finally, the establishment of populations at restored sites is likely to be ruled by complex interactions between tolerant species already present at a site, arriving targeted species and possibly also arriving invasive alien species not targeted by restoration. These biotic interactions are poorly understood and currently render prediction of restoration effects on species level almost impossible (Feld et al. 2011).

Another important confounding factor is the scale of restoration in rivers.The scale issue is very often undrestimated. Stretches that are restored are often too short for processes to recover and pressures and stressors are often acting at a much larger scale than the scale of the restoration.

Lakes

Eutrophication

The capacity for a lake to recover to its original state is not only dependent upon a sufficient reduction in the primary pressure, but also on the occurrence of secondary pressures that may confound the recovery process. An overview of all primary and secondary drivers and pressures reported in 364 peer-reviewed manuscripts was given in Table 2. In 302 lakes (lake-equivalent recovery case studies) eutrophication was the primary pressure and in 45 lakes also a secondary pressures was reported (Figure 26).

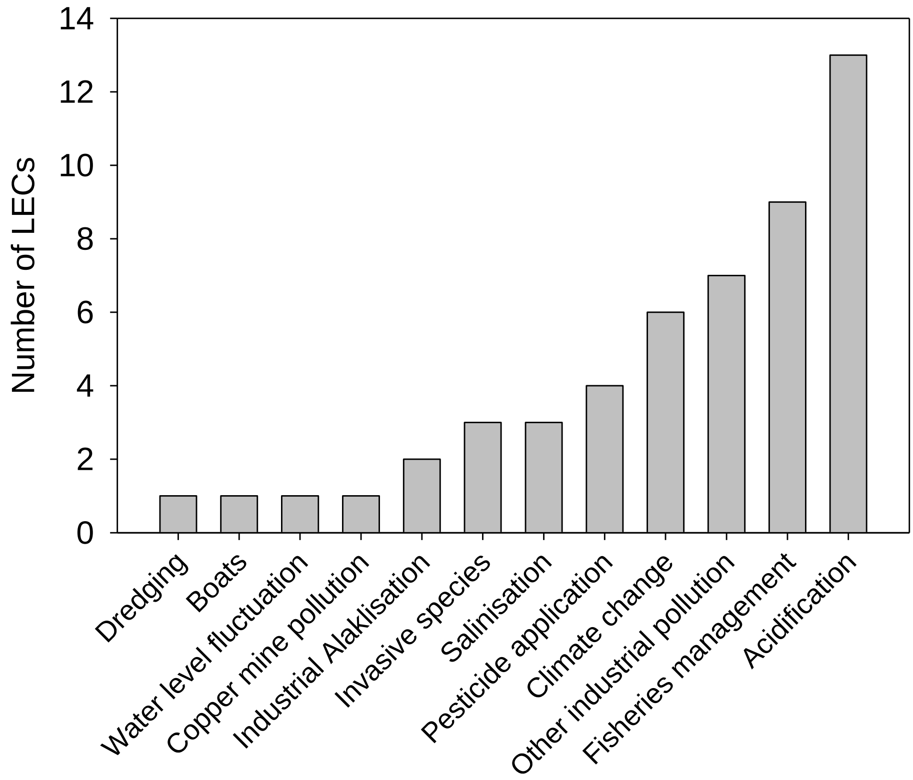


Figure 26. Number of lake equivalent case studies (LECs) reporting each of the secondary pressures shown along the horizontal axis in combination with eutrophication.

Acidification, fisheries management, industrial pollution and climate change were the main secondary pressures impacting on nutrient enriched lakes and, with the exception of industrial pollution which covers a wide range of independent pollutants and practices, these are discussed below. Surprisingly, the role of invasive non-native species was rarely reported as a secondary pressure affecting ecological responses to eutrophication control in lakes, even though invasive species (especially macrophytes) were commonly reported to occur during the recovery process. It was concluded that multi-pressure scenarios may have been under represented in the literature for a number of reasons, including:

- secondary pressures were not considered to alter the impact of the target BQE and, therefore, were not reported,
- secondary pressures were not yet identified at the study site, and
- no data existed with which the secondary pressures could be quantified.

In contrast, a wide range of secondary pressures have been reported at Loch Leven (Scotland), a shallow lake that is undergoing recovery from eutrophication following a significant (60%) reduction in external nutrient load (May and Carvalho, 2010). It is likely that each of the pressures has an impact on every BQE at this site at some level, either directly or indirectly, through complex feedbacks within the system. Although current research aims to quantify the relationships between pressures and the BQEs in Loch Leven, no single article has reported on all of these pressures in the context of lake recovery from eutrophication. Schindler (2006) reviewed a range of factors known to confound the recovery of lakes from eutrophication and stressed the need for better understanding of multiple pressures and identified the following secondary pressures as being of particular importance: (1) the aggravation of eutrophication by climate warming, (2) the overexploitation of piscivorous fishes and (3) changes in silica (Si) supply from the catchment as a result of climate change. Schindler (2006) hypothesised that the latter could lead to a reduction in Si concentrations in lakes leading to cyanobacteria outcompeting diatoms within the phytoplankton community. Some other recovery confounding factors are acidification, fisheries management, and non-native species.

Although lake acidification is expected to alter the recovery process and end-point biological communities in lakes following eutrophication management, few studies report specifically on this type of potentially confounding impact at the whole lake scale. Clearly, soft water lakes are most sensitive to acidification pressures and inferences can be made with respect to the likely impacts based on available literature. The effects of acidification on the phytoplankton are generally considered to be a switch in community composition from chrysophytes, cryptophytes and diatoms to non-N_2 fixing cyanophytes or to dinoflagellates (Blomqvist, et al., 1993). However, the combined effects of acidification and eutrophication on phytoplankton community structure are still contested within the literature, with experimental studies showing either an increase, or no change, in community structure as eutrophication and acidification increases (Irfanullah & Moss, 2005; Reynolds et al., 1998). The impacts of acidification on macrophytes in soft water lakes with a conductivity 1-2 meq l^{-1} are well reported (Arts, 2002; Brouwer & Roelofs, 2001). In general, acidification causes a switch from an unimpacted community dominated by acid intolerant soft water macrophytes (to a deteriorated end point that was characterised by loss of all submerged macrophytes coupled with the development of *Sphagnum*, other bryophytes, *Juncus bulbosus* and filamentous epiphytic and benthic algae. A study by Havens et al. (1993) suggests that, in acidified lakes, the zooplankton community will become dominated by smaller zooplankton and, therefore, the shift from small to large bodied zooplankton observed during recovery from eutrophication may be unbalanced.

Fisheries management practices vary in scale from fish stocking to support recreational fishing in small lakes to industrial fishing in large lakes. Any alteration to the fish community can result in top-down impacts on lower trophic levels through an increase or reduction in grazing pressures (Carpenter & Kitchell, 1996).

Non-native invasive species are defined by the GB non-native species secretariat as “any non-native animal or plant that has the ability to spread causing damage to the environment, the economy, our health and the way we live.” Many non-native invasive species are introduced by humans for a specific purpose (e.g. aquaculture; Figure 17). However, by definition, they also have the ability to spread *via* transport pathways throughout the environment and infest lakes. This spread may operate with and without human intervention. A summary of the types of non-native invasive species important in freshwater lakes and their vectors of infestation is outlined in Table 11. Depending on the invasive species lake recovery maybe either enhanced or hampered. Some examples are:

- Rosenthal et al. (2006) reported a decrease in the germination of macrophyte species as a result of invasion by crayfish;
- Matsuzaki et al., (2009) reported a decrease in macroinvertebrate abundance in the presence of common carp;
- among others, impacts of invasive dreissenid mussels (e.g. Zebra mussel; *Dreissena polymorpha*) include a reduction in phytoplankton biomass (up to 45% for more than 7-10 years post invasion; Higgins et al., 2011) and total P concentrations in the water column (Hecky et al., 2004).

Table 11. Summary of organism groups, mode of introduction, environmental impacts and examples for non-native invasive species considered to pose a high risk to the recovery of lakes from eutrophication (Manchester & Bullock, 2000).

Type of organism	Purpose	Impacts	Examples
Fish and shelfish	Angling, accidental introduction	Competition, predaition, habitat disturbance, disease vector, increased sediment P release, reduction of macrophyte cover	Grass carp, common carp, crayfish
Invertebrates	Accidental introduction	Competition with native species, grazing of phytoplankton	Zebra mussel, alien gammarids
Plants	Accidental introduction	Prolific vegetative growth, forms dense matts leading to deoxygenation, outcompetes native flora.	*Crassula helmsii, Egeria densa* and *E. Nutalii*

Biomanipulation

Søndergaard et al. (2007) concluded that in general it is difficult to draw firm conclusions about the reasons for unsuccessful restorations using biomanipulation. Apart from insufficient reduction of the external phosphorus loading, a number of different internal mechanisms have been suggested to contribute to the limited sustainability of restoration effects (Table 12). Although the focus of the study by Søndergaard et al. (2007) was on biomanipulation, the study also included other internal lake restoration measures as listed in Table 12.

Table 12. 'Internal' reasons for failure (excluding insufficient external nutrient loading reduction) of lake restoration in Denmark and the Netherlands from Søndergaard et al. 2007).

Method	Reason
Fish removal	Insufficient number of fish removed Rapid return of strong cohorts of zooplanktivorous fish Invertebrate predators (*Neomysis*/*Leptodora*) reduce the zooplankton High resuspension rate of loose sediment Internal P loading because of formerly high external loading 'Instability' because of low coverage of submerged macrophytes
Pike stocking	Low survival of stocked fish, for example because of predation and cannibalism

	Low pike consumption of young-of-the-year fish Bad timing of pike stocking relative to the hatching of young-of-the-year cyprinids
Sediment removal	Low P sorption capacity of new sediment surface Incomplete dredging
P-fixation	'Ageing' of alum and reduced P retention capacity Reduction/binding of ferric chloride by carbonate or sulphide
Oxygenation	No permanent effects achieved? Continued oxygenation Increased mobile phosphorus pool because of mineralization

Insufficient external loading reduction, internal phosphorus loading and absence of stable submerged macrophyte communities to stabilize the clear-water state are, according to Søndergaard et al., 2007, the most probable causes for a relapse to earlier conditions after initial recovery. Gulati et al. (2008) mention bottle necks similar to those mentioned by Søndergaard et al. (2007) and put them in a schematic overview (Figure 27). In many cases it was not possible to adequately reduce the available P in lake water resulting both from excessive external and internal (release from sediment) P inputs (Gulati et al. 2008). Thus, it might be necessary to reduce P load to a ''biomanipulation efficiency threshold of P-loading'' which may be in the range of 0.6–0.8 g TP m_2 yr_1 as hypothesised by Benndorf and Miersch (1991). Only then a sustained reduction of phytoplankton via top-down control might be possible due to both direct (grazing) and indirect (top-down induced P reduction) mechanisms (Benndorf et al., 2002). Such a 'threshold' level of P loading may however vary from lake to lake. Thus, both insufficient reduction of external P load to the lake after the restoration measures, and an increased rate of P release from the lake sediment into the overlying water would be crucial factors in offsetting the success of measures.

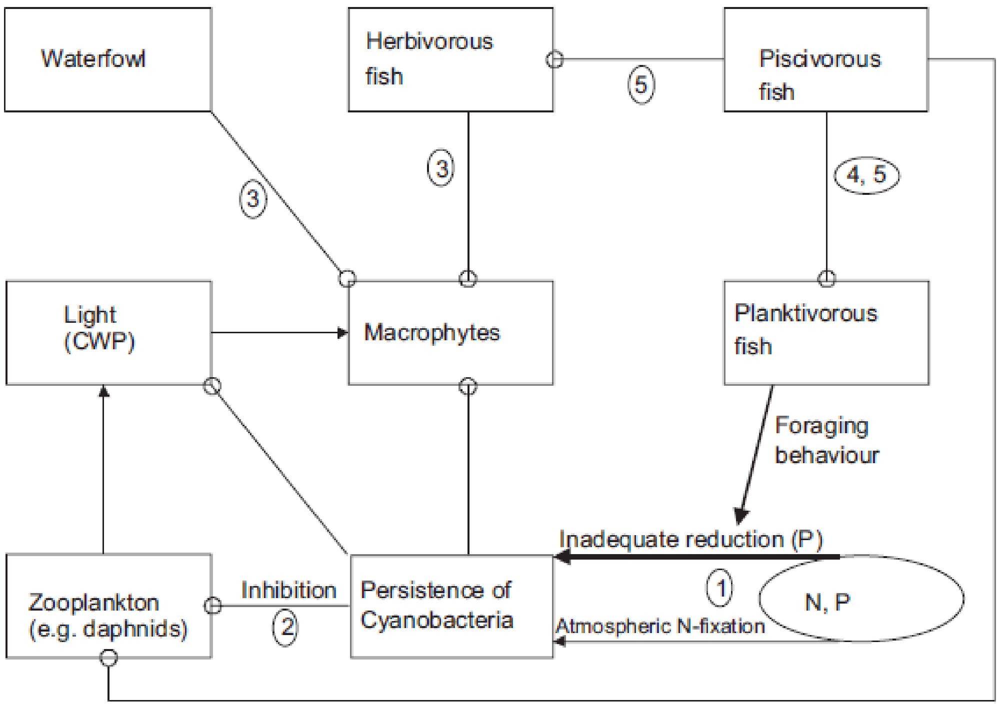


Figure 27. Schematic view of the bottlenecks (indicated from 1 to 5 small circles) in food webs, that prevent the success of biomanipulation measures in the restoration of shallow lakes. They are: (1) the inadequate reduction of allochthonous P and the increase of autochthonous P, (2) the poor edibility of filamentous and colonial cyanobacteria to zooplankton, (3) an inadequate coverage of the lake area by macrophytes due to predation by both waterfowl and herbivorous fish, (4) the ineffective reduction of planktivorous fish biomass and the inability to maintain it at low levels for longer periods, and (5) the failure of introduction of northern pike to develop a population at the level that can control planktivorous fish. Lines with black arrow heads represent a positive influence of one parameter on the other (e.g. zooplankton grazing has a positive effect on light). Lines with white circles represent a negative (inhibitory) influence of one parameter on another (e.g. waterfowl depress macrophytes through predation). CWP in box for light = clear water phase, N = nitrogen, P = phosphorus. The lines and arrows from Planktivorous fish to P and from N,P to Persistence of Cyanobacteria are thicker than other lines because nutrient reduction and fish removal are the most severe bottlenecks (after Gulati et al. 2008).

Thanks to biomanipulation research, we now know that macrophytes play a crucial role in maintaining long, clear-water periods in lakes (Carpenter and Lodge 1986, Jeppesen et al. 1990, Gulati & Van Donk 2002, Hosper et al. 2005, Gulati et al. 2008). Gulati & Van Donk (2002) presented a schematic representation (Figure 28) of the mechanisms and factors causing sediment resuspension and turbidity in shallow lakes in relation to macrophytes (submerged plants). In 25 of these 34 lakes (see also Søndergaard et al. 2007), where the macrophyte development was monitored, 18 lakes showed no increase in percentage of macrophyte cover. Thus, the improvement in light climate may not per se be attributed to increase in macrophyte biomass.

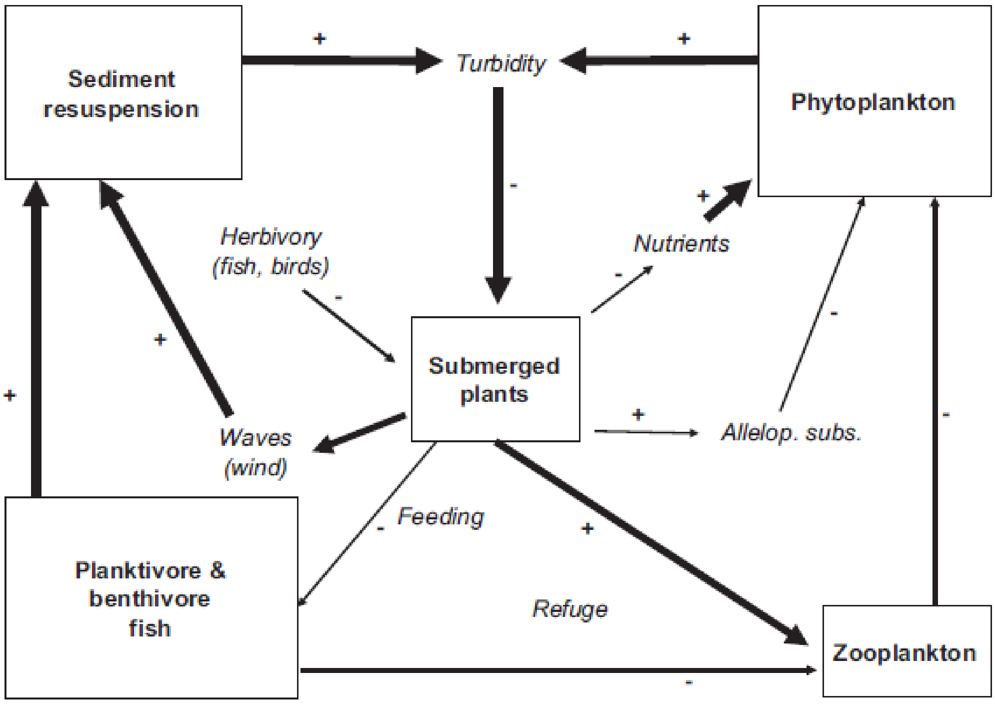


Figure 28. Schematic representation of the mechanisms and factors causing sediment resuspension and turbidity in shallow lakes in relation to macrophytes (submerged plants). After lake restoration the increase in macrophytes plays an important role in reducing sediment resuspension and turbidity and improving the underwater light climate. Feed-back mechanisms and their strength are indicated with arrows (after Gulati & Van Donk 2002). (Allel. Subs.= allopathic substances)

Acidification

Although acidification affects aquatic biota in a predictable way (Økland and Økland 1986, Appelberg and Degerman 1991), the underlying processes and mechanisms governing recovery are not always clearly understood (Hildrew and Ormerod 1995, Strong and Robinson 2004). Many factors, such as habitat connectivity, dispersal abilities, food availability, and species interactions, might individually or collectively affect the rates and trajectories (e.g. hysteresis) of biological recovery. According to Yan et al. (2003), three main bottlenecks could impede biological recovery: 1) inadequate water quality, 2) insufficient supply of colonists, and 3) community-level interactions. Detection of biological recovery also might depend on the choice of indicator (chemical, biological) and the choice of habitat (stream, lake; pelagic, benthic) (Johnson et al. 2006; Johnson and Hering 2009). Furthermore, many site-specific factors, such as latitude, altitude, catchment characteristics (e.g., soil/bedrock type, land use/cover), ecosystem size, and nutrient status, might result in lag responses that differ among sites, and thereby confound attempts to detect change, resulting in high levels of uncertainty.

Few studies have addressed the stability of aquatic communities following liming, despite the fact that the ultimate goal of liming is to restore ecosystems to their pre-disturbed conditions

(Appelberg 1995). Explanations for lack of long-term recovery have focussed on food web complexity, impediments to recolonization and establishment (e.g. fish), variability in environmental conditions resulting from liming, and biotic interactions. In a recent study of long-term effects of liming on lake assemblages, Angeler and Goedkoop (2010) found that associations between functional feeding groups indicated less connectivity and food web complexity in limed lakes relative to circumneutral and acidified lake types. These authors speculated that repeated lime applications comprise frequent pulse disturbances which offset the establishment of stable trophic relationships in the food webs of limed lakes.

A review of 81 papers from the peer-reviewed literature included an examination of the factors hindering or preventing recovery from acidification. Most papers reported that chemical recovery had taken place following deposition reductions although there were exceptions. A lack of chemical recovery was ascribed to insufficient reduction of sulphur deposition, the effects of nitrogen deposition, soil acidification and increases dissolved organic carbon (one paper in each case) with two papers each highlighting the acid episodes and failure of liming measures. In many more cases limited or no biological recovery was reported. These were divided between abiotic and biotic constraints and are listed below together with the number of papers where each is cited.

<u>Abiotic contraints</u>
Nutrient nitrogen (6 papers), acid episodes (6), toxic metals (3), UV (3), site characteristics (6), increases in DOC (3), climate (8) and calcium (3)

<u>Biotic constraints</u>
Community closure (6), recolonisation (8), decoupled food-web (7), functional shifts (2), within-species adapatation (1), absence of fish predation (6), stable simplified food-web (2), competetive resistence (4)

Estuarine and coastal waters

Much less references were available on failure or delay of recovery in estuarine and coastal waters. Borja et al. (2006) suggested that contaminants in contact with oxic water, after reducing eutrophication or organic pollution, can be released back into solution (Calmano et al. 1993), causing toxic effects in the biota. For example, Trannum et al. (2004) have shown that high copper levels in sediments (400 to 1500 mg kg^{-1}) had a distinct negative effect on benthic colonisation. This observation would facilitate the management implementation of the extremely costly remedial action plans to remove ‘hot spots’ of sediment contamination; otherwise, such hot spots could delay or impede faunal recovery once dissolved oxygen conditions in the estuary have improved (Sáiz-Salinas & González Oreja 2000, González Oreja & Sáiz-Salinas 2003, Gorostiaga et al. 2004).

Carstensen et al. (2011) noted that the response of chlorophyl-a to changing nitrogen conditions differs between individual coastal areas. The authors suggest several ecosystem features that could potentially account for this, e.g. differences in tidal ranges, secchi-depth, mixing and the fraction of refractory TN. This suggests that ecosystem characteristics can play an important role in the outcome of restoration projects. The same authors also suggested that shifting baseline (as a result of global change), may explain the reported failure to revert eutrophied coastal ecosystems to their previous state following reduction of nutrient inputs.

Summary

Several major reasons return in many publications on recovery failure or delay:

- Spatial scale must be large enough (catchment).
- Temporal scale: there is time needed for recovery.
- Multistressors present: mostly only one or a few stressor were tackled, others forgotten.
- Confounding abiotic processes affect recovery, such as upstream ‘hidden’ stressors, internal P loading, and biological interactions, like the early arrival of non-native species, but also climate change effects, effects of management and maintenance.
- Distance from source populations and lack of connectivity results in dispersal limitations and colonisation barriers.
- There is no guiding monitoring that makes evaluation along the development and redirection of measures possible.

Chapter 9. Recovery: Shifting baselines

Rivers

Shifting baselines imply that the present state of the system is not an adequate reference to evaluate the effectiveness of restoration effort, as the future status of the ecosystem would differ from that at the present under a 'do nothing' scenario.

In all peer reviewed literature examined no references were made to shifting baselines nor to (quantified) thresholds.

Lakes

The concept of regime shifts was apparent within the lakes although none of the studies specifically set out to quantitatively assess response trajectories. The general approach was instead to assess specific responses over a short time scale. The main reason for failure of restoration in the case studies appeared to be insufficient control of catchment or internal TP loading or through lack of sustained control of fish stocks. In terms of acidification there are very few studies (and none included in the review) which address the concept of shifting baselines. However, recent (unpublished) data show how the recovery trajectories in some lakes are not tracking back towards the species communities found at the equivalent stage of the degradation phase. Several reasons have been proposed for this including the effects of atmospherically deposited nitrogen acting as a nutrient in N limited systems and the effects of climate change driven increases in temperature. Although there is little direct evidence that climate change has altered baselines so that new system equilibria have resulted, independent of the effects of existing pressures, increasing numbers of studies have identified cases where, despite measures taken to combat the effects acidification, the recovery trajectory is not indicating a return to pre-impact reference conditions.

Estuarine and coastal waters

Carstensen et al. (2011) found a parallel trend towards increase in chlorophyl-a yield per unit nitrogen in the past decade in all regions examined and they indicate this could be the result of the major shift in the baselines for the functioning of coastal ecosystems resulting from the combined effect of climate change, overfishing and, possibly, other components of global change. The shift in the functional relationship between chlorophyl-a and TN over time reported by Carstensen et al. (2011) helps to explain reported failure to revert eutrophied coastal ecosystems to their previous state following reduction of nutrient input (Duarte et al. 2009). Despite observed increases in chlorophyl-a concentrations it is still important to stress that nutrient do release pressure on the ecosystem and improve conditions relative to what these would have been under a ‘do nothing scenario’. The Nervión estuary is an example of shifting baselines. In this estuary, recovery occurred with decreasing pressures but the system did not return to its original state, since many ecosystems had been reduced or lost (e.g. intertidal areas, salt-marshes, etc.). Hence, restoration has provided a new or ‘alternative stable state’ for the Nervión estuary, and the net result will probably lead to a decreased abundance, richness and biomass of some biological taxa (Borja et al. 2010).

Summary

It is difficult to judge whether the concept of shifting baselines is part of the reality of ecosystems developments as proof is hard to find. Even in the coastal and estuarine examples it is questionable whether the responses are due to alternative states or due to overlooked other stressors. Often in many lake examples the latter is the case.

Chapter 10. Recovery: Effects of biological interactions

Scale and biological interactions

As restoration efforts usually occur on a small scale (i.e. habitats), tailored to the important scale for target species, and ecological processes subject to restoration on a large scale, there is a discrepancy between the two (Bult et al. 1998; Jansson et al. 2007, Lake et al. 2007). This so called scoping problem can lead to a mismatch between the requirements of individual species and the scale at which habitat or connections are restored. Many scientists therefore stress the need of large scale restoration efforts (Bond & Lake 2003, Lake et al. 2007, Palmer 2009), advocating whole watershed and whole estuary restoration efforts. At the same time often socioeconomic constraints (resources and conflicts of interest) will limit projects on a large scale.

Processes related to metapopulation dynamics, dispersal and connectivity play a role on different scales and the observed scale of importance is depending on scope of the study. For example frequency of dispersal of invertebrates among lakes, depends upon perspective and spatial scale (Havel & Medley 2006): When two adjacent lakes are observed, dispersal might seem limited, since a single species is occurring in one lake, whilst being absent in another. On the other hand when genetic markers are used to study a global set of lakes, this same species is rapidly invading reservoirs in new regions indicating nearly unlimited dispersal capability.

A second example is the scale at which dispersal and colonization processes in stream ecosystems are important for restoration. Within stream colonization of aquatic invertebrates through drift is a well-studied phenomenon (Townsend & Hildrew 1976). On a larger scale, i.e. catchment or even across catchments, other processes (e.g. insect flight) become important and these are less frequently studied.

Freshwater ecosystems are typically embedded in a heterogeneous landscape and restoration sites are not only influenced by activities in the target area, but also by processes around it (Wiens 2002). Apart from ecological processes the right scale of habitat is important as well. For many species habitat size is of central importance and largely predicts the viability of the metapopulation (Bond & Lake 2003). Therefore, even if correct types of habitat are restored, but at the wrong scale, still restoration may not be effective.

In restoration ecology another scaling issue occurs, when knowledge is to be extrapolated to a different scale. It might not be possible to extrapolate knowledge on abiotic factors known to work on a specific scale to a different scale since assumptions and generalizations are scale specific. Therefore, it is difficult to incorporate all the required information at the right scale that is needed to predict the processes by which restored ecosystems develop.

Determining the right scale for species, the extent of the required (different) habitats, the scale a which processes like dispersal and connectivity occur in the context of restoration and even the scale at which large-scale processes that eventually influence habitat quality is an important aspect of successful restoration. However the spatial scales that are most important often remain poorly understood.

With regard to recovery from acidification, there are two keys measures operating at very different scales. Reducing the source of the acidification through emission controls provides the circumstances whereby 'natural' recovery can take place and acts at a regional scale. Liming is a site specific measure that accelerates the timescale of the potential recovery but does not necessarily mimic the natural recovery process. The bulk of the evidence to date indicates that biological recovery may be is or will be delayed following chemical restoration. The notion under acid conditions ecological niches vacated by acid-sensitive species can subsequently be reinvaded by the same taxa when water chemistry is restored is over simplistic. The process of biological recovery in chemically recovered freshwaters is not well understood and is subject to hysteresis and stochasticity. Recovery rates and trajectories can be influenced by many different factors, both biotic and abiotic. These include dispersal constraints (although some studies have ruled out inter-basin dispersal and habitat as limits on recovery) and sporadic acid episodes. The role of or ecological inertia leading to biological resistance to recolonisation has been examined. The trophic structure of stream communities is changed by acidification and there is evidence that acid-tolerant species fill the ecological roles of their acid-sensitive counterparts. This ‘community closure’ results in niches abandoned by sensitive species (for example the loss of sensitive grazing species) being subsequently filled by more tolerant taxa (for example acid tolerant shredding generalists), potentially obstructing the route to reinvasion. Thus internal shifts in function (e.g. acid tolerant detritivores operating as herbivores replacing lost grazers) can provide some biological resistence to recovery in terms of species composition. Biological resistence can also lead to delays in the recovery of aquatic food webs when there are differential rates of recolonisation. For example zooplankton community structure can remain impacted following chemical recovery if lakes remain fishless due to low colonization rates, resulting in a lack of top down control on macroinvertebrate predators.

Dispersal, connectivity and establishment

As outlined restoration projects often deal with more than one factor important for restoration outcome. In many systems both the lack of landscape connectivity and sufficient local propagule sources severely limit the regeneration of native species in degraded communities. Causes of declines of source populations of native species can be habitat destruction and fragmentation, which will limit the effectiveness of regional pools as a source of propagules for recolonisation. This, combined with the absence of native species in the degraded site and the loss of a native propagule bank, limit the regenerative ability of many native species in restoration projects (Shurin et al. 1994). So in order to encourage re-establishment, not only local factors should be restored, but measures to improve the quality of source populations or enhance connectivity should be incorporated in restoration effort as well.

Restoration success is dependent on the possibility of populations to colonize the new and restored habitat. Due to knowledge gaps and scale discrepancies, both habitat and dispersal constraints still restrict restoration outcome in many programmes (Lake et al. 2007). For example, since most dispersal is known to occur over short distances and linearly in streams, instream barriers can largely constrain the possibilities of colonization (Bond & Lake 2003). A study by Blakely et al. (2006) revealed formerly unrecognized physical barriers to aquatic insect colonization in urban streams. Road culverts acted as partial barriers to upstream flight with 2.5 x more individuals occurring downstream of road culverts than upstream. Apart from 'hard barriers', like dams and weirs, soft barriers are associated with the isolation of restored habitat. This isolation might be related purely to distance of the restored site to the source population, to unfavourable intervening habitat or to dispersal constraints of the target species of interest, resulting in unlikely colonisation of the restored habitat (Bond & Lake 2003). Even without dispersal constraints, colonization that leads to actual establishment might be restricted due to insufficient habitat quality (habitat limitation or establishment limitation) (e.g. minimal habitat size, lack of oviposition sites) (Blakely et al. 2006).

Non-native species

Freshwater systems are suggested to be particularly prone to invasions by alien species, as they are utilised intensively by people in ways that maximise opportunities for spread and establishment of invaders (Schreiber et al. 2002). Invasion of exotic species after restoration may play a role in the restoration process in the different ways (Table 13). Due to their life history characterization as being both good colonizers after disturbance and persistent community members, exotic species may respond rapidly to habitat restoration and are likely to return after removal (D'Antonio & Meyerson 2002), thereby outcompeting native biota (Bond & Lake 2003). Additionally, although native species often become demographically vulnerable as a result of habitat fragmentation, invasive or otherwise undesirable species are often well established in degraded lands. These species can thereby establish a dominant position in degraded systems and, thus, management efforts can have the unintended effect of facilitating the spread of these species (van Riel et al. 2006). Furthermore, formerly isolated ecosystems could become connected due to restoration measures that enhance connectivity leading to introductions exotic species. Often these exotic species are the first species to arrive, possibly indicating a life history advantage. Once introduced invading species can largely alter the (a)biotics of a restored site.

Table 13. Summary of organism groups, mode of introduction, environmental impacts and examples for non-native invasive species considered to pose a high risk to the recovery of lakes from eutrophication (Manchester & Bullock, 2000).

Type of organism	Purpose	Impacts	Examples
Fish and shelfish	Angling, accidental introduction	Competition, predaition, habitat disturbance, disease	Grass carp, common carp, crayfish

		vector, increased sediment P release, reduction of macrophyte cover	
Invertebrates	Accidental introduction	Competition with native species, grazing of phytoplankton	Zebra mussel, alien gammarids
Plants	Accidental introduction	Prolific vegetative growth, forms dense matts leading to deoxygenation, outcompetes native flora.	*Crassula helmsii, Egeria densa* and *E. Nutalii*

Invading species have been shown to compete for resources (food and space) with native species. Furthermore, they can change the recipient environment by acting as an ecosystem engineer, or even by providing habitat for native species (Schreiber et al. 2002). Examples of connectivity restoration leading to unintended spread of invasive species are numerous (Kondolf et al. 2006). For example, Paillex et al. (2009) focused on the macroinvertebrate response related to the increases in lateral connectivity of secondary channels. The effects of an increase in the hydrologic connectivity on the biological characteristics of macroinvertebrate assemblages were assessed with a model indicating gradient maximum colonization potential in the most connected channels. Nevertheless, the post-restoration sampling showed a large proportion of colonizers were favoured by the restoration operations and non-native species occurred in the restored channels. In their recommendations the scientists state that in restoration projects a diversification of the hydrologic connectivity of channels is to be preferred over maximum hydrologic connectivity (Paillex, 2009).

Another example describes the invasion of zebra mussel *Dreissena polymorpha* to formerly isolated basins that were connected via canals in North American Great Lakes. Zebra mussel initially invaded via ballast water, and subsequently invaded through canals and lakes, as did some species of fish (Mills et al. 1993). Additionally, inter-catchment connections in Australia caused formerly isolated fish species to become introduced to new watersheds, thereby threatening local species' existence (Lintermans 2004).

In some recent cases, naturally connected systems are being intentionally fragmented to prevent or hamper introduction of undesirable invasive fish species. For example, the use of dams to prevent sea lampreys from reaching spawning grounds in Great Lake tributary streams (Porto et al. 1999). Another example is the availability of only small scour pools for native fish, too small to support larger-bodied exotic invaders (Bond & Lake 2003).

Although most effects of invading species are classified as negative and unwanted, some studies indicate that invading species might positively influence native species (Rodriguez 2006). For example, in a study by Schreiber et al. (2002) experiments were carried out in which the densities of the invading aquatic snail *P. antipodarum*: were experimentally manipulated. Results showed that there were no negative effects and even a positive relationship between *P.*

antipodarum densities and native fauna abundance and densities was observed. More in general, such positive mechanisms include habitat modification, trophic subsidy, pollination, competitive release, and predatory release (Rodriguez 2006).

In the context of restoration both negative and facilitating effects of invading species should be integrated, either to minimise invasions and their impacts or to modify the expected outcome of restoration (Jannsson et al. 2007). Furthermore, connectivity is not always good, neither always bad since like all changes, connectivity is likely to benefit some organisms at the expense of others (Kondolf et al. 2006).

Other biological aspects

For macroinvertebrates the recovery is dependent on the constituent species, which have different life-cycles, reproduction periods and patterns of larval dispersal. The dispersal of reproductive stages of macroinvertebrates, due to storms in coastal areas or high flows in streams or wave action in lakes, may hamper the recovery of this BQE. Recovery of fish populations was found to be dependent on the previous recovery of benthic communities on which they feed. In their review for recovery processes of marine animal populations (with focus on long-lived mammals, birds, reptiles and fishes) and ecosystems, Lotze et al. (2011) highlighted the several factors from which recovery depends (Table 14). For example, long-lived animals have lower intrinsic rates of increase and will take longer to rebound to higher abundance levels than will short-live species (Lotze et al. 2011).

Table 14. Population factors from which recovery depends (after Lotze et al. 2011).

Population factors	Ecosystem health	Diversity
Life history (time to rebound)	Water quality	Genetic diversity
Magnitude of depletion and Allee effects	Habitat availability	Species richness
Habitat range and occupancy	Species interactions	
Population structure (e.g. juvenile:adult, male:female and meta-population)	Primary production	
	Climate	

The habitat and range of occurrence of a species might also have a role. Recovering species often occupy a greater percentage of their historic range compared with non-recovering species (Abbitt and Scott, J.M., 2001). An example is given by Lewison, R.L. et al. (2004) whom observed that more coastal marine mammals and sea turtles have shown more recovery than have offshore species, which face continued threats and less management

Summary

Restoring the appropriate habitat is still the main component of aquatic ecosystem restoration efforts. Although the importance of establishing the suitable abiotics is stressed by a multitude of studies, the awareness that other factors should be considered as well is apparent in recent recommendations on freshwater restoration (Bond & Lake 2003, Jansson et al. 2007). There are several, more or less connected issues that are repeatedly stressed in a multitude of studies:

1) Incorporating the spatial and temporal scale (i.e. maximum and minimum) of the habitat and the connectivity between the various habitat patches, including both abiotic and biotic components;

2) Incorporating the knowledge of source populations and dispersal ability or constraints in predicting restoration outcome. However few studies attempt to match this ecological background with empirical data.

3) Incorporating mitigating measures to prevent non-native species to colonise and set priority effects.

Most restoration projects incorporate the restoration of abiotic conditions to a fixed end-point, i.e. an ideal average condition. However, communities tend to be shaped by abiotic extremes and restoration planning should be shaped according to these extremes. Re-colonisation of a species is only likely when the entire scope, i.e. the maximum and minimum spatial extent and temporal duration of habitat use is restored. Furthermore, extreme events will amplify the importance of the presence of refugia. Along with habitat enhancement, restoring refugia, in this way enhancing the resistance and resilience to both natural and anthropogenic disturbances, may be critical to survival and colonisation of target populations. Subject to many studies is the importance of biological factors in determining ecosystem structure and function, providing both habitat structure and biological interactions that shape community build up. In order to create suitable habitat for some species other species, that affect the focal species' habitat, need to be involved in restoration efforts and planning. Furthermore the restoration target often includes community build up, indicating the importance of incorporating the role of biological interactions in restoration planning. This includes accounting for a long restoration period that may be needed to restore ecosystem function, including biotic factors. Finally, our main aim was to construct a driver – pressure – state – impact – recovery chain for the biological processes of metacommunity dynamics and connectivity. As such, most review studies suggest that there is a lack of quantitative data with which driver-impact relationships can be described. This is apparent within the relatively short post-management monitoring periods for the majority of restoration projects.

Information on dispersal, connectivity and metapopulation dynamics lacks in almost all monitoring data. Especially, dispersal is extraordinarily difficult to study (Macdonald et al. 2002) and methods to study dispersal have many restrictions even when a combination of techniques (direct and indirect) is used. In addition these techniques are mostly used to study

dispersal and connectivity in existing ecosystems and are hardly ever used in restoration projects. Furthermore, conclusions from studies on the effects of dispersal constraints and connectivity constraints are almost never firm and hard to extrapolate to the restoration practice, since results are confounded by the effects of environmental constraints, which cannot be excluded.

Chapter 11. Recovery: Impacts of climate and global change

Rivers

Observed climate change over the last century is influencing European streams and rivers in many ways. It affects running water ecosystems directly as well as indirectly through societal and economic systems, such as agricultural practices and land-use. In many cases, climate change is an additional stress factor. Biodiversity, for example, is primarily affected by factors such as land-use changes, overexploitation of natural resources, invasive alien species, and air pollution. But the role of climate change is expected to become more dominant, in particular if the magnitude and rate of climate change is at the higher end of the projected range (EEA 2004; Solomon et al. 2007).

The European Environment Agency (2004) summarised the impacts of Europe's changing climate and listed three key messages for running waters:

- Annual river discharge has changed over the past few decades across Europe. In some regions, including eastern Europe, it has increased, while it has fallen in others, including southern Europe. Some of these changes can be attributed to observed changes in precipitation.
- The combined effect of projected changes in precipitation and temperature will in most cases amplify the changes in annual river discharge.
- Annual discharge is projected to decline strongly in southern and south-eastern Europe but to increase in almost all parts of northern and north-eastern Europe, with consequences for water availability.

The hydrologic consequences all over Europe of most climate model results show a precipitation decrease in summer and an increase in autumn-winter. Furthermore, extreme daily (especially summer storm) precipitation may become more frequent (Räisänen et al. 2003). Consequently, discharge may show a more fluctuating regime with more extremes (higher spates and longer droughts), and lower predictability of the annual discharge regime (Arnell 1999).

A key concern, especially for stream ecosystems, is if and how such climate-induced hydrological regime change will influence river channel morphology, channel and riparian habitats, and species diversity. In large parts of Europe hydromorphological alterations, such as channel straightening, weir and dam construction, disconnection of the river from its floodplain and alteration of riparian vegetation, are major stressors affecting streams and rivers (Kristensen & Hanson 1994; Armitage & Pardo 1995; Hansen et al. 1998). This poses the question what additional role future climate conditions will play upon current and future land and water uses.

The climate induced changes in flow regime relate to four features, namely (i) high flows and spates, (ii) low flows and droughts, (iii) rate of flow change or flow dynamics, and (iv) annual shift in the flow pattern. The effects of spates and droughts both depend on the magnitude, frequency, duration, timing and rate of change (Poff et al. 1997; Gasith & Resh 1999). The main hydromorphological effects of high flows and spates are:

- Scouring of accumulated sediment and debris.
- Redistribution of streambed substrate and organic matter in the channel.
- Changing channel morphology and forming new erosion (runs and riffles) and deposition (point and mid-channel bars, pools, sand accumulations) zones.
- Washing away in-channel and encroaching riparian vegetation.
- Homogenizing water quality conditions along the stream channel and adjacent water bodies.
- Increased shear stress on organisms.

The major hydromorphological effects of low flows and droughts are:

- Siltation of fine mineral and organic material.
- Decrease in oxygen content and an increase of nutrients and minerals.
- Mineralization of organic material in the stream bottom.
- Drying of the banks, reducing their stability.
- Absence of water.
- Drought stress on organisms.

Lakes

The North Atlantic Oscillation (NAO) index is used as a proxy for weather conditions, especially in North-Western Europe, and is calculated by comparing patterns of sea level atmospheric pressures near the "Icelandic low" and the "Azores high" of the North Atlantic Ocean (Jones et al. 1997, Jones et al. 2003). The main climatic variations associated with the NAO are warmer temperatures from late-autumn to early-spring and stronger, more westerly winds during years of strong positive NAO values (Slonosky et al. 2000). Past studies have identified NAO as a good (although variable; Gerten & Adrian 2001) indicator of climatic forcing of ecological and physical processes in European lakes (reviewed by Straile et al. 2003). Some case studies showed that the NAO could confound lake recovery when external loads were reduced by generating increased run-off and, consequently, higher nutrient inputs from external sources. Coherent positive correlations have been observed between the indices of the NAO and surface and hypolimnetic lake water temperatures (Livingstone 1999, Livingstone 2000, Livingstone & Dokulil 2001, Dokulil et al. 2006) as well as lake chemistry variables (Monteith et al. 2000; Evans et al. 2001; Weyhenmeyer 2004) across a range of European lake types. These drivers have, in turn, been linked to alterations in the ecological signatures of lakes, most strikingly affecting plankton dynamics and the onset of the clear-water phase (Gerten & Adrian, 2000; George, 2000; Straile, 2002).

In shallow lakes, the temperature effects of the NAO have been shown to drive ecosystem functioning through the regulation of steady state change (Scheffer et al. 2001, Rip 2007). Regulation of wind intensity and direction may also have direct impacts on the depth of wind induced turbulence and, therefore, habitat disturbance in the littoral zones of lakes (Spears & Jones 2010).

The occurrence of extreme weather events is expected to increase as a result of climate change. Shallow lakes appear to be particularly sensitive to such events, with changes in ecological state being triggered by hurricanes (Havens et al. 2001, Scheffer 1998), high intensity rainfall (Rip 2007, Nöges et al. 2010), drought leading to water level decrease (Coops et al. 2003), severe winter ice events leading to anoxia and winter fish kills (Scheffer 1998), and heat waves (Schindler 2006).

Jeppesen et al. (2005) mention that the confounding effects effect of global warming have been examined for some lakes included in their study (e.g. Straile & Adrian 2000, Anneville et al. 2002, Kangur et al. 2002, Nöges et al. 2004). According to Jeppesen et al. (2005): "These analyses indicate an earlier onset of the clear-water phase (if any), stratification (if any) and fish spawning, reduced mixing in stratified lakes, and higher surface water temperature promoting higher internal P loading from sediment portions exposed to warm surface water. Moreover, in shallow and some deep lakes cyanobacteria may be more abundant and blooms may persist longer. However, the strong re-oligotrophication signals revealed by our analysis suggest that the observed changes in the lakes included in our data set reflect primarily the impacts of lower nutrient loadings rather than climate change. This conclusion is supported by results from mesocosm experiments which likewise suggest a much stronger effect of changing nutrient loadings than of changing temperatures in shallow lakes (McKee et al. 2003, Moss et al. 2003)." In short, effects of global change are likely to run counter to reductions in nutrient loading rather than reinforcing re-oligotrophication. The general re-oligotrophication response patterns can be regarded only as a guideline when discussing the response of a particular lake. Each lake is unique in many respects and may exhibit a specific trajectory (Jeppesen 2005, Moss et al. 2005). In a warming climate, it is likely that N removal processes will be enhanced, potentially leading to an increase in the resilience of cyanobacteria to oligotrophication measures (Schindler 2006, Wehenmeyer 2007, Wehenmeyer et al. 2008). However, N loading may also be enhanced under future climate change scenarios with regional-scale uncertainty in climate predictions leading to uncertainty in the expectations of future N status in lakes (Jeppesen et al. 2011).

The trends in monthly dissolved inorganic N (DIN: i.e. NH_4-N+NO_3-N+NO_2-N) concentration in many recovering lakes indicate a reduction in DIN from winter through to late summer-early autumn (Figure 12). This reduction has been attributed to a combination of denitrification and biological uptake, both of which are expected to increase with temperature (Jensen et al. 1992, Van Donk et al. 1993).

There has been recent controversy about the role of N- and P-limitation in lakes and estuaries and the effectiveness of reducing only one of these nutrients to improve water quality (Pearl 2009, Schindler et al. 2008). In a Policy Forum Review in the journal Science, Conley et al.

(2009) concluded that effective control of the negative impacts of nutrient enrichment ought to involve controlling and reducing the availability of both nutrients.

The potential influence of climate change on surface-water acidity is now a central focus of research, and studies in remote mountaiom regions have shown that temperature was probably the key driver of pH in the period before industrialization and also that periods of increased pH in the twentieth century could be attributed to climate warming, despite a trend towards decreasing pH from acid deposition. (Psenner & Schmidt, 1992; Sommaruga-Wögrath et al., 1997). As sulphur deposition continues to decrease, temperature may emerge as the strongest driver of lake acidity at many sites. However, the potential effects of changing precipitation patterns are also likely to be important as the increased precipitation and storminess predicted for some mid- to high-latitude regions may cause more acid conditions in future through higher stream-flow episodes (cf. Evans *et al.* 2008). In addition increased drought frequency may delay the recovery of acidified lakes through drying and oxidation of sulphur-polluted catchment wetlands (cf. Faulkenham *et al.* 2003).

The NAO is an important regional-level driver of climate in southern Sweden (Weyhenmeyer 2004). Positive NAO values signify higher precipitation and, thus, more variable hydraulic and chemical conditions. Johnson and Angeler (2010) showed that littoral invertebrate assemblages responded more readily to the NAOwinter index than did phytoplankton assemblages. Life-history characteristics mediate the strength of biological responses to climate forcing (Adrian et al. 2006). Much shorter generation times of phytoplankton relative to invertebrates could uncouple phytoplankton responses from climate signals and could make these responses visible only via indirect or integrated effects (Ottersen et al. 2001). These effects could be manifested through responses to abiotic or biotic variables that are more strongly affected by the NAO. Rusak et al. (2008) showed climate imprints on zooplankton dynamics, and these effects could have at least some influence on phytoplankton. Durance and Ormerod (2007) found that the imprints of the NAO on stream invertebrate assemblages were particularly strong in circumneutral streams. They suggested that acidification stress in streams could override a climate signal, possibly because of the complex interaction of the natural and anthropogenic disturbance regimes to which stream organisms are exposed under the constraints set by climatic conditions. The difference between the study of Durance and Ormerod (2007) and Johnson and Angeler (2010) could be because lake ecosystems are much less affected by changes in surface-water hydrology than are stream ecosystems, and hence, stream invertebrate assemblages respond more readily to climatic variability regardless of acidification status.

Estuarine and coastal waters

The following text is fully based on Harley et al. (2006). Given their global importance, estuarine and coastal environments are a major focus of concern regarding the potential impacts of anthropogenic climate change. The basic predictions can be summarized as follows: as temperature rises in the future, the distribution and abundance of species will shift according to their thermal tolerance and ability to adapt. However, a growing body of work is demonstrating that these simplistic relationships between temperature and the biota are inadequate in predicting many important aspects of future biological change. Patterns of temperature change in space and time, and biological responses to them, are not as straightforward as once envisioned. More importantly, temperature is only one of a suite of potentially interacting climatic variables that will drive future ecological change in marine systems. Finally, studies conducted on population- and community-level processes suggest that climatic impacts on individual organisms do not necessarily translate directly into changes in distribution and abundance (Harley et al. 2006; Figure 29). Anthropogenic climatic forcing is mediated primarily by greenhouse gas (predominantly CO_2) emissions. Together, elevated CO_2 and the resultant increases in global mean temperature will result in a cascade of physical and chemical changes in marine systems. Because warming trends will be stronger over continental interiors than over oceans, the atmospheric pressure gradient, and thus wind fields, along ocean margins will intensify. Stronger wind fields might lead to enhanced upwelling in eastern boundary currents (Bakun 1990), which could increase nutrient availability at the surface. Changes in atmospheric circulation might also change storm frequency; an increase in the frequency of winter storms has already been observed in coastal oceans (Bromirski et al. 2003), and the trend is expected to continue (IPCC 2001). Continued uptake of atmospheric CO_2 is expected to substantially decrease oceanic pH over the next few centuries, changing the saturation horizons of aragonite, calcite, and other minerals essential to calcifying organisms (Kleypas et al. 1999; Feely et al. 2004). Model estimates of pH reduction in the surface ocean range from 0.3 to 0.5 units over the next 100 years and from 0.3 to 1.4 units over the next 300 years, depending on the CO2 emission scenario used (Caldeira & Wickett 2005).

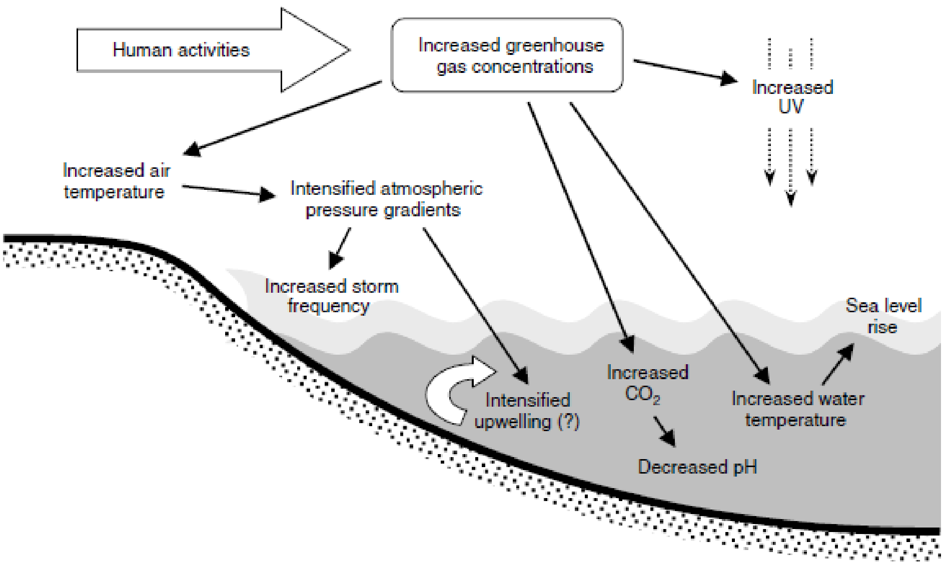


Figure 29. Important abiotic changes associated with climate change. Human activities such as fossil fuel burning and deforestation lead to higher concentrations of greenhouse gases in the atmosphere, which in turn leads to a suite of physical and chemical changes in coastal oceans.

The question mark indicates that the relationship between climate change and upwelling is uncertain (after Harley et al. 2006).

Proximal ecological responses to changing environmental conditions are:

- Responses to temperature
- Responses to sea level rise
- Responses to changes in circulation
- Responses to CO_2 and pH change
- Responses to UV

Emergent ecological responses comprise:

- Distributional shifts: zonation patterns
- Distributional shifts: biogeographical ranges
- Changes in species composition, diversity and community structure
- Changes in primary and secondary production
- Changes in population dynamics and evolution

Because stable populations and intact communities appear to be more resilient to climatic disturbances such as episodic heat waves and storms, such protective measures may help to minimize the risk of population collapses, community disruption, and biodiversity loss (Hughes et al. 2003). The designation of protected areas should be based at least in part on known spatial and temporal refuges that can act as buffers against climate-related stress (Allison et al. 1998). Fisheries managers must also incorporate climate change into consideration when determining fishery management plans (Jurado-Molina & Livingston 2002).

Summary

The direct effects of climate change on ecosystems (Figure 30) impact the performance of individuals at various stages in their life history cycle via changes in physiology, morphology and behaviour. Climate impacts also occur at the population level via changes in transport processes that influence dispersal and recruitment. Community-level effects are mediated by interacting species (e.g. predators, competitors, etc.), and include climate-driven changes in both the abundance and the per capita interaction strength of these species. The combination of these proximate impacts (upper box) result in emergent ecological responses (lower oval), which include alterations in species distributions, biodiversity, productivity and microevolutionary processes.

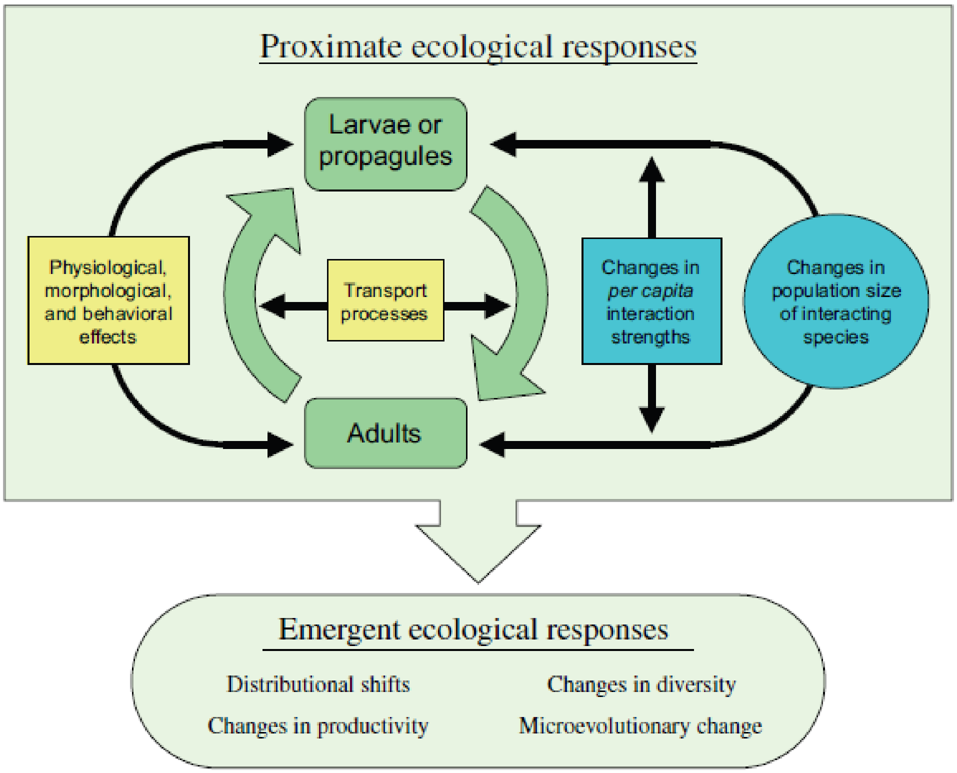


Figure 30. Potential ecological responses to climate change. The life cycle of a generic species is shown in green. Abiotic changes in the environment have direct impacts (yellow boxes) on dispersal and recruitment, and on individual performance at various stages in the life cycle. Additional effects are felt at the community level via changes in the population size and per capita effects of interacting species (in blue). The proximate ecological effects of climate change thus include shifts in the performance of individuals, the dynamics of populations, and the structure of communities. Taken together, these proximate effects lead to emergent patterns such as changes in species distributions, biodiversity, productivity, and microevolutionary processes (after Harley et al. 2006).

A range of biological management practices (especially fishery management) and extreme weather events were identified as key factors that were responsible for slowing down or contradicting lake recovery processes. In contrast, the loss of dissolved nitrogen (N) through denitrification and biological uptake, leading to a switch from P- to N-limitation of primary production in summer/autumn, was identified as a potential recovery enhancing process. Alterations in nutrient concentrations and biogeochemical cycling at the sediment-water interface, following nutrient management, can influence the magnitude and timing of nutrient delivery to downstream ecosystems. This phenomenon is likely to be highly sensitive to changes in local weather conditions associated with climate change.

Chapter 11. Research gaps

Rivers

In summary, there is sufficient evidence to develop best-practice guidance for riparian buffer restoration and related in-stream habitat improvements, but there is only weak quantifiable evidence for statistical or mechanistic relationships as a basis for quantifying and modelling the effects of restoration and biological recovery.

Lack of long-term monitoring

Restoration measures should be monitored beyond the time-scale of typical experimental (PhD) studies, i.e. >3–4 years, in order to detect long-term recovery, but also adverse effects. The current knowledge on long-term restoration effects is scarce. Given the long history of river basin degradation in large areas of Europe, ecologists, restorationists and water managers probably need to be very patient, until better-designed and integrated restoration schemes will show the desired effects in the future. Nevertheless, only a sufficiently frequent and long-term monitoring scheme will help provide more insight into the spatial and temporal effects of restoration. This knowledge is likely to be the key to design effective and successful ecological restoration schemes in the future.

Lack of data on phytobenthos

Irrespective of the type of restoration measure studied. Aquatic phytobenthos were less frequently addressed in the literature. Macrophyte data mainly for dam removal and re-meandering for in-stream mesohabitat enhancement and riparian buffers data on macrophytes are also sparse.

Lakes

Lack of statistical understanding of ecological responses

There is currently a lack of knowledge on the uncertainty surrounding ecological responses following specific management practices. This is due in part to the use of multiple management measures and a lack of data (see next bullet). A more targeted management approach would lead to more scientifically sound case studies. In this respect, standard lake management guidance documentation (i.e. ‘decision support systems’) at the EU level may result in a more useful data set in the longer-term that is relevant to WISER targets and BQE metrics. Some of the lakes had sufficient data to conduct statistical analyses (e.g. ‘before/after’) but did not do so. In the short-term, a re-analysis of these data using a common statistical procedure would improve our understanding of the effectiveness of specific management scenarios.

Need more comprehensive and long-term monitoring to underpin quantitative assessment of management measures

Many studies report recovery time scales based on monitoring scenarios that are very short. Measures should be taken to conduct rigorous scientific assessments of lakes where single management options have been conducted. This could involve a space for time approach where ecological communities are compared across lakes with similar management histories (e.g. Jeppesen et al. 2005 for external load reduction).

Need to quantitatively assess driver-impact relationships during the recovery process.

Most of the studies in which statistics were used determined whether there was or was not a response in an indicator variable following a specific management scenario. However, little effort was given to understanding the interactions and synergies between indicator variables. In the first instance, models (e.g. PCLAKE, PROTECH) may be used to form a set of testable hypotheses for a series of lakes in which management has been conducted before the collection of contemporary data with which these theories can be tested. Specific areas of interest would include (1) tipping points and regime shifts, (2) overriding pressures (e.g. temperature, wind etc), (3) the role of ecosystem structure on resilience to restoration, and (4) commonality in response trajectories across trophic levels for specific lake types and management scenarios. The summarised BQE responses can also be used as hypotheses in this regard.

Lack of case studies relevant to WFD TP targets

The majority of the lakes reported post-management TP concentrations in excess of the WFD TP targets (i.e. > 0.5 mg TP l-1). As such, it is unlikely that a meta-analysis of such case studies will provide a comparison to expected recovery trajectories within the TP range of WFD targets. For example, Spears et al. (2011) summarised the expected recovery trajectory of the phytoplankton community with decreasing TP concentration. A large number of case studies did not achieve post-management TP concentrations below 0.1 mg TP l-1, the effective concentration for required changes in the phytoplankton community.

Lack of data on macroinvertebrates and fish.

Very few case studies report data with which the macroinvertebrate and fish community responses can be assessed quantitatively. In the case of macroinvertebrates, this appears to be due to a lack of studies concentrating on littoral or profundal community shifts at any time scale. In the case of fish, case studies reporting fish data commonly have had biomanipulation work conducted which confounds the assessment of ‘natural’ recovery trajectories.

Biomanipulation

Many difficulties arise when interpreting the results obtained from lake restoration projects, and fundamentally lake restoration still involves a large proportion of trial and error, where the mechanisms for a successful restoration remain largely unclarified. Carpenter *et al*. (2001) similarly found difficulty in predicting the conditions under which food web structure will control pelagic primary producers. Restorations are often conducted primarily to improve water quality and are not designed as a scientific experiment (Mehner *et al*. 2002). This implies that multiple restoration measures are often used more or less simultaneously, rendering it impossible to disentangle fully the impact of individual measures (amongst others Søndergaard et al. 2007).

Estuarine and coastal waters

Lack of long-term monitoring

There are very few examples of long-term monitoring data, including different biological elements (i.e. plankton,benthos, fishes, etc.) together with physico-chemical data from waters and sediments, showing the recovery trajectories after remediation or restoration processes in marine environments (see examples in Borja et al. 2009, Elliott et al. 2007, Jones & Schmitz 2009, Lotze et al. 2006, Simenstad et al. 2006, Stein and Cadien 2009, Yuksek et al. 2006) to address this fundamental gap in our knowledge about recovery patterns and rates in estuarine and coastal ecosystems (Borja et al. 2010).

Lack of information on the most important factor(s) for recovery

Multiple interacting forces are driving recovery in marine and estuarine ecosystems (e.g. reduction of cumulative human impacts or any combination of reduced threats with favorable environmental, ecological, social or economic factors (Lotze et al. 2011)), but little information exists on which combination of factors is the most important for recovery.

Lack of knowledge on shifting baselines

Concerning the setting of conservation and management targets, it is crucial to identify historical reference points, the carrying capacity for individual populations and ecosystems, and to assess how changes in ecosystems or environmental conditions over time have altered such baselines and, hence, recovery prospects (Lotze et al. 2011). Provided the importance of shifting baselines for the setting and evaluation of actions to reverse eutrophication, it is fundamental that our understanding of the causes of such shifts in baselines improves to allow forecasting the trajectories of individual coastal ecosystems. A better understanding of the dynamics of coastal ecosystems forced by both changes in nutrient inputs, derived from factors operating at the basin scale, and shifting baselines derived from forces operating at various scales is fundamental to achieve this goal. To gain this better understanding there is a need for physiological experiments, experiments at the mesocosm scale, and large scale experiments conducted at ecosystem level and sustained over long time scales, supported by modelling efforts (Carstensen et al. 2011).

Summary

In summary, there is need for the following research efforts;

- Need for statistical understanding of ecological responses.
- Need for more comprehensive and long-term monitoring to underpin quantitative assessment of management measures.
- Need to quantitatively assess cause-effect relationships during the recovery process.
- Need for case studies relevant to WFD targets.
- Need for specific knowledge on certain BQEs in certain water categories.
- Need for knowledge on maintenance, and recurring management.
- Need for knowledge on the most important factor(s) for recovery and their interactions.
- Need for knowledge on shifting baselines and thresholds.

Chapter 12. Conclusions

The main drivers of eutrophication, acidification and hydromorphological degradation are population growth resulting continuous increases in urbanisation (changes in flows of water run-off and of nutrients and other substances), industrialisation (air pollution/acidification and flows of substances), land use (agricultural intensification affecting flows of water, landscape morphology and run-off of substances) and water use changes (e.g., drinking water, recreation). These drivers are related to a wide range of pressures.

The key surface water pressures are related to agricultural land use (e.g., drainage run -off, water inlet, organic waste and fertiliser inflow, salinisation, soil erosion and losses), discharges from industry (e.g., acidification, waste and nutrient inflow, loading with heavy metals and others toxic components, detergents and soaps, inflow of cooling water), urbanisation (e.g., waste water treatment works, drainage networks, housing, paved surface and road run off, introduction of invasive species) and water use (e.g., water level management, fishery management, boating (sediment disturbance), sediment dredging, macrophyte harvesting).

The Driver-Pressure-State-Impact-Response-Recovery (DPSIRR) scheme provides a framework to link socio-economy with ecology. Literature was searched for existing DPSIRR-chains for the three water categories. Such conceptual models on the recovery of river, lake and estuarine and coastal ecosystems were scarce and fragmented. Such models lacked for the marine systems were quite one-sided, focusing on eutrophication, for lakes and quite specific for certain measures in rivers. Comparison and integration of DPSIRR-chains is up date impossible.

There is a common agreement that drivers and pressures in general are the same in lakes, rivers and estuarine and coastal waters. From the selection and availability of literature it is though clear that eutrophication and acidification got most attention in lake studies, hydromorphological changes were the focus of river studies and recovery studies in estuarine and coastal marine waters were limited and diverse in drivers and pressures studied.

Although a multitude of studies provide theoretical frameworks, guidelines, research needs and issues that are important for freshwater restoration, only few studies provide evidence of how this ecological knowledge might enhance restoration success. Goals of restoration projects typically encompass a multitude of objectives (species groups, ecological, cultural and landscape values) and a multitude of measures. Thus, evaluation of the response of a single factor to a single measure tends to be difficult (Roni et al. 2008).

In rivers most measures target the morphology of the stream stretch or the instream habitats. Few only are related to reduction of nutrient input. On the contrary, in lakes all measures target to reduce nutrient levels, especially phosphate. Others mainly focus on acidification. Measures are not often taken directly in estuarine and coastal waters, these much more relate to measures taken inland through legislation on nutrient reduction. These observations supported our initial hypothesis that “at a catchment scale, nutrient stress affecting functional (production/decomposition) processes will be more important in lakes and marine systems, while hydromorphological stress affecting habitat availability will be more important in rivers”.

Another major bottleneck is the lack of sufficient monitoring (Palmer et al. 2005), allowing for insufficient learning from both successful and unsuccessful restorations (Jansson et al. 2007, Palmer 2009). However, the frequently occurring general recommendation in proposed guidelines for restoration projects (Palmer et al. 2005, Perrow & Davy 2002), including appropriate monitoring and publishing of the results, could help to gain insight into the processes important to successful restoration.

A third problem is related to the many detected effects that occur only in the short-term and at the local (site) scale, which raises the question of appropriate scaling for restoration. There is not yet evidence for the most appropriate spatial nor temporal scale, but several extended review studies supported the hypothesis that the local scale is inappropriate to achieve long-term measurable improvements. Local restoration measures are often ‘swamped’ by larger scale, e.g. in rivers the reach scale or in rivers and lakes the watershed scale pressures that continue to affect the treated sites. Such limitations imply that the spatial scaling of restoration schemes must fit the scaling of degradation, e.g. the scale of the stressors impacting the system.

In rivers and lakes quite an amount of monitoring data are available. In estuarine and coastal waters such data are scarce. Despite the number of monitored recovery cases, each one seems to stand alone as monitoring schemes were set-up for local situations and to answer partial questions. Furthermore, in many, many cases data on recovery just lack and this is quite alarming! Not only is the amount of available data surprisingly low, the composition of the available data is often very limited and does not allow the evaluation and generalisations of improvements and eventually of successes. The huge investments in recovery of surface waters require control of the ecological effects. Therefore, restoration monitoring should become mandatory. Only by frequent monitoring of biological and abiotic changes after restoration will restoration practitioners and scientist be able to evaluate the success of the restoration measure and eventually of the investment done.

The majority of restoration studies in rivers and in estuarine and coastal ecosystems have focused on macroinvertebrates. In rivers also fish are important indicators. In lakes phytoplankton is the BQE studied most extensively. The difference in indicator groups used goes back to the primary causes of degradation. In lakes eutrophication is most important and phytoplankton best reflects the nutrient status of the lake over time. In rivers most degradation goes with hydromorphological change. Macroinvertebrates and fish respond strongly tot these types of changes. The choice of macroinvertebrates as indicators of degradation in estuarine and coastal waters is less obvious as eutrophication and organic load are most common causes of degradation along with bottom disturbances. The latter would best be reflected in macroinvertebrate responses the first less. The confounding factor in estuarine and coastal waters for phytoplankton is water movement. Water movement reduces the indicative value of phytoplankton.

Although, analyses in the different reviews do not address full recovery’, authors do give indications on ‘full recovery’ based on estimates. Marine ecosystems may take between 35 and 50 years to recover. Recovery after weir removal may take as long as 80 years. Recovery after riparian buffer installment may take at least 30-40 years. Despite the fact that they do not indicate ,‘full recovery‘ we compared recovery times between the three water categories as mentioned in the different reviews. In marine ecosystems benthic invertebrates and macrophytes have the potential to recover within months (in two studies on recovery of sediment disposal) and fish within one year. When only marine studies that recover from eutrophication are included, recovery times for macroinvertebrates varied between >3 years and >6 years. Although in some cases recovery can take <5 years, especially for the short-lived and high-turnover biological components, full recovery of estuarine and coastal ecosystems from over a century of degradation can take a minimum of 15–25 years for attainment of the original biotic composition, diversity and complete functioning may lag far beyond that period. In lakes recovery time from eutrophication for macroinvertebrates varied between 10 and 20 years. As in marine ecosystems recovery of macrophytes (2 to >40 years) and fish in lakes (2 to >10 years) be relatively fast. Response times for organism groups in rivers are lacking, because the literature rarely includes post hoc monitoring of more than 5 years. Also, the fact if biological response in rivers occurs within short term is undecided. The potential benefits of most in-stream structures will be short-lived (<10 years) unless coupled with riparian planting or other process-based restoration activities supporting long-term recovery of key ecological and physical processes.

In both rivers and lakes the success rate of restoration measures appears to be much higher for the abiotic conditions than for the biotic indicators. Since eutrophication is considered to be the most important pressure in rivers and lakes, only this is not addressed in rivers, this might be a major cause. Especially, the response of macroinvertebrates in rivers is questionable, some studies mention recovery times of others question recovery of macroinvertebrates completely. In lakes internal nutrient loading often delays recovery.

Several major reason return in many publications on recovery failure or delay:

- Spatial scale must be large enough (catchment).
- Temporal scale: there is time needed for recovery.
- Multistressors present: mostly only one or a few stressor were tackled, others forgotten.
- Confounding abiotic processes affect recovery, such as upstream ‘hidden’ stressors, internal P loading, and biological interactions, like the early arrival of non-native species, but also climate change effects, effects of management and maintenance.
- Distance from source populations and lack of connectivity results in dispersal limitations and colonisation barriers.
- There is no guiding monitoring that makes evaluation along the development and redirection of measures possible.

It is difficult to judge whether the concept of shifting baselines is part of the reality of ecosystems developments as proof is hard to find. Even in the coastal and estuarine examples it is questionable whether the responses are due to alternative states or due to overlooked other stressors. Often in many lake examples the latter is the case.

Restoring the appropriate habitat is still the main component of aquatic ecosystem restoration efforts. Although the importance of establishing the suitable abiotic conditions is stressed by a multitude of studies, the awareness that other factors should be considered as well is apparent in recent recommendations on freshwater restoration. There are several, more or less connected issues that are repeatedly stressed in a multitude of studies:

- Incorporating the spatial and temporal scale (i.e. maximum and minimum) of the habitat and the connectivity between the various habitat patches, including both abiotic and biotic components;
- Incorporating the knowledge of source populations and dispersal ability or constraints in predicting restoration outcome. However few studies attempt to match this ecological background with empirical data.
- Incorporating mitigating measures to prevent non-native species to colonise and set priority effects.

A range of biological management practices (especially fishery management) and extreme weather events were identified as key factors that were responsible for slowing down or contradicting recovery processes. Alterations in nutrient concentrations and biogeochemical cycling at the sediment-water interface, following nutrient management, can influence the magnitude and timing of nutrient delivery to downstream ecosystems. This phenomenon is likely to be highly sensitive to changes in local weather conditions associated with climate change.

In summary, there is need for the following research efforts;

- Need for statistical understanding of ecological responses.
- Need for more comprehensive and long-term monitoring to underpin quantitative assessment of management measures.
- Need to quantitatively assess cause-effect relationships during the recovery process.
- Need for case studies relevant to WFD targets.
- Need for specific knowledge on certain BQEs in certain water categories.
- Need for knowledge on maintenance, and recurring management.
- Need for knowledge on the most important factor(s) for recovery and their interactions.
- Need for knowledge on shifting baselines and thresholds.

In conclusion, restoration ecology is just in its infancy. The huge amount of literature evaluated brings up one major conclusion. Restoration is a site, time and organism group specific activity. Generalisations on recovery processes are up to date hard to make. Despite the multitude of studies that provided theoretical frameworks, guidelines, research needs and issues that are important for freshwater restoration, only few studies provide evidence of how this ecological knowledge might enhance restoration success.

References

Abbitt, R.J.F. and Scott, J.F. 2011. Examining differences between recovered and declining endangered species. Cons. Biol. 15: 1274-1284.
Adrian, R., S. Wilhelm, and D. Gerten. 2006. Life-history traits of lake plankton species may govern their phenology response to climate warming. Global Change Biology 12: 652–661.
Allan, J.D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35: 257–284.
Allan, J.D., Erickson D.L. and Fay, J. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshw. Biol. 37: 149-161.
Alewell, C., M. Armbruster, J. Bittersohl, C. D. Evans, H. Meesenburg, K.Moritz, AND A. Prechtel. 2001. Are there signs of acidification reversal in freshwaters of the low mountain ranges in Germany? Hydrology and Earth System Sciences 5: 367–378.
Allison, G.W., Lubchenco, J. and Carr, M.H. 1998. Marine reserves are necessary but not sufficient for marine conservation. Ecol. Appl. 8: S79–S92.
Angeler D.G. and Goedkoop, W. 2010. Biological responses to liming in boreal lake: an assessment using phytoplankton, macroinvertebrate and fish communities. Journal of Applied Ecology 47: 478-486.
Anneville, O., Gammeter, S.and Straile, D. 2005. Phosphorus decrease and climate variability: mediators of synchrony in phytoplankton changes among European peri-alpine lakes. Freshwater Biology 50: 1731-1746.
Appelberg, M. and Degerman, E. 1991. Development andstability of fish assemblages after lime treatment.Canadian Journal of Fisheries and Aquatic Sciences 48: 546–554.
Appelberg, M., Henrikson, B.-I., Henrikson, L. & Svedäng, M. (1993) Biotic interactions within the littoral community of Swedish forest lakes during acidification. Ambio 22: 290–297.
Appelberg, M. and Svensson, T. 2001 Long-term ecological effects of liming – The ISELAWprogramme.Water, Air and Soil Pollution, 130: 1745–1750.
Armitage, P.D. & Pardo, I. (1995) Impact assessment of regulation at the reach level using macroinvertebrate information from mesohabitats. Regulated Rivers: Research & Management 10, 147-158.
Arnell, N.W. (1999) The effect of climate on hydrological regimes in Europe: a continental perspective. Global Environmental Change 9, 5-23.
Arts, G.H.P. 2002. Deterioration of Atlantic soft water macrophyte communities by acidification, eutrophication and alkalinisation. Aquatic Botany 73: 373-393.
Baillie, B.R., Garrett, L.G. and Evanson, A.W. 2008. Spatial distribution and influence of large woody debris in an old-growth forest river system, New Zealand. For. Ecol. Manage. 256: 20–27.
Barton, D.R., Taylor, W.D. and Biette, R.M. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in Southern Ontario streams. North Am. J. Fish. Manage. 5: 364–378.
Bednarek, A.T. 2001. Undamming rivers: A review of the ecological impacts of dam removal. Environ. Manage. 27(6): 803–814.
Beechie, T.J., Sear, D.A., Olden, J.D., Pess, G.R., Buffington, J.M., Moir, H., Roni, P. and Pollock, M.M. 2010. Process-based principles for restoring river ecosystems. BioScience 60: 209–222.
Benndorf, J. and Miersch, U. 1991. Phosphorus loading and efficiency of biomanipulation. Verh./Int. Ver. Theor. Angew. Limnol. 24: 2482–2488.

Benndorf, J., Böing, W., Koop, J. and Neubauer, I. 2002. Top-down control of phytoplankton: the role of time scale, lake depth and trophic state. Freshwat. Biol. 47: 2282-2295.
Bergquist, B. 1995. Supplementary measures to aquatic liming. Pp 399-418, In: Liming of acidified surface waters: A Swedish synthesis (Henriksen and Brodin, eds). Springer.
Bernhardt, E.S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Powell, B. & Sudduth, E. 2005. Synthesizing U.S. River Restoration Efforts. Science 308: 636–637.
Blakely, T.J., Harding, J.S., McIntosh, A.R. and Winterbourn, M.J. 2006. Barriers to recovery of aquatic insect communities in an urban stream. Freshwater Biology 51: 1634–1645.
Blocksom, K.A. and Flotemersch, J.E. 2008. Field and laboratory performance characteristics of a new protocol for sampling riverine macroinvertebrate assemblages. River Research and Applications 24(4): 373-387.
Blomqvist, P., Bell, R.T., Olofsson, H., Stensdotter, U. and Vrede, K. 1993. Pelagic ecosystem responses to nutrient additions in acidified and limed lakes in Sweden. Ambio 22: 283-289.
Bond, N.R. and Lake, P.S. 2003. Local habitat restoration in streams: constraints on the effectiveness of restoration for stream biota. Ecol. Manage. Restor. 4: 193–198.
Borja, A, Bald, J., Franco, J., Laretta, J., Muxika, I., Revilla, M., Rodriguez, J.G., Solaun, O., Uriarte, A., Valencia, V. 2009. Using multiple ecosystem components, in assessing ecological status in Spanish (Basque Country) Atlantic marine waters. Marine Pollution Bulletin 59: 54-64.
Borja, A., Dauer, D.M., Elliott, M. and Simenstad, C.A. 2010. Medium- and long-term recovery of estuarine and coastal ecosystems: patterns, rates and restoration effectiveness. Estuaries and Coasts 33: 1249–1260.
Borja, A., Muxika, I. and Franco, J.2006. Long-term recovery of soft-bottom benthos following urban and industrial sewage treatment in the Nervión estuary (southern Bay of Biscay). Marine Ecology Progress Series 313: 43–55.
Bourque, C.P.A. and Pomeroy, J.H. 2001. Effects of forest harvesting on summer stream temperatures in New Brunswick, Canada: an inter-catchment, multiple-year comparison. Hydrol. Earth Syst. Sci. 5: 599-613.
Bromirski, P.D., Flick, R.E. and Cayan, D.R. 2003. Storminess variability along the California coast: 1858–2000. J. Climate 16: 82–993.
Brouwer, E. & Roelofs, J.G.M. 2001. Degraded soft water lakes: Possibilities for restoration. Restoration Ecology 9: 155-166.
Bult, T.P., Haedrich, R.L. and Schneider, D.C. 1998. New technique describing spatial scaling and habitat selection in riverine habitats. Regulated Rivers: Research & Management 14: 107–118.
Burkhead, N.M. and Jelks, H.L. 2001. Effects of suspended sediment on the reproductive success of the tricolor shiner, a crevice-spawning minnow. Trans. Am. Fish. Soc. 130: 959-968.
Caldeira, K. and Wickett, M.E. 2003. Anthropogenic carbon and ocean pH. Nature 425: 365.
Calmano, W., Hong J. and Förstner, U. 1993. Binding and mobilization of heavy metals in contaminated sediments affected by pH and redox potential. Water Sci Technol 28: 223–235.
Carpenter, S.R. and Kitchell, J.F. 1996. The trophic cascade in lakes. Cambridge University Press, UK. pp 385.
Carpenter, S.R. and Lodge, D.M. 1986. Effects of submerged macrophytes on ecosystem processes. Aquat. Bot. 26: 341–370.
Carpenter, S.R., Caraco, N.F., Howarth, R.W., Sharpley, A.N. and Smith, V.H. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol. Appl. 8: 559-568.

Carpenter, S.R., Cole, J.J., Hodgson, J.R., Kitchell, J.F., Pace, M.L., Bade, D., Cottingham, K.L., Essington, T.E., Houser, J.N. and Schindler, D.E. 2001 Trophic cascades, nutrients, and lake productivity: whole-lake experiments. Ecological Monographs 71: 163–186.
Carstensen J., Sánchez-Camacho, M., Duarte, C.M., Krause-Jensen, D. and Marbà, N. 2011. Connecting the dots: responses of coastal ecosystems to changing nutrient concentrations. Environmental Science and Technology 45(21): 9122–9132.
Castelle, A.J., Johnson, A.W. and Conolly, C. (1994) Wetland and stream buffer size requirements - a review. J. Environ. Qual. 23: 878–882.
Clements, W.H., Carlisle D.M., Lazorchak, J.M. and Johnson, P.C. 2000. Heavy metals structure benthic communities in Colorado mountain streams. Ecol. Appl. 10: 626-638.
Cloern, J.E. 2001 Our evolving conceptual model of the coastal eutrophication problem. Mar. Ecol. Prog. Ser. 210: 223–253.
Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C.and Likens,, G.E. 2009. ECOLOGY Controlling Eutrophication: Nitrogen and Phosphorus. Science 323: 1014-1015.
Cooper, C.M. 1993. Biological effects of agriculturally derived surface water pollutants on aquatic systems-a review. Journal of Environmental Quality 22: 402-408.
Coops, H., Beklioglu, M.and Crisman, T.L. 2003. The role of water-level fluctuations in shallow lake ecosystems - workshop conclusions. Hydrobiologia 506: 23-27.
D'Antonio, C. and Meyerson, L.A. 2002. Exotic Plant Species as Problems and Solutions in Ecological Restoration: A Synthesis. Restor. Ecol. 10: 703-713.
Davies-Colley, R.J. and Quinn, J.M. 1998. Stream lighting in five regions of North Island, New Zealand: Control by channel size and riparian vegetation. N. Z. J. Mar. Freshwat. Res. 32: 591–605.
Davies-Colley, R.J., Meleason, M.A., Hall, G.M. and Rutherford, J.C. 2009. Modelling the time course of shade, temperature, and wood recovery in streams with riparian forest restoration. N. Z. J. Mar. Freshwat. Res. 43: 673–688.
Degerman E. and Appelberg, M. 1992. The response of stream-dwelling fish to liming. Environmental Pollution, 78: 149-155.
Delong, M.D. and Brusven, M.A. 1998. Macroinvertebrate communitys tructure along the longitudinal gradient of an agriculturally impacted stream. Environ. Manag. 22: 445-457.
Diaz, R. and Rosenberg, R. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321: 926–929.
Didderen K. & Verdonschot P.F.M. 2010. Biological processes of connectivity and metapopulation dynamics in aquatic ecosystem restoration subtitle. Wageningen, Alterra, Alterra. Deliverable 6.4.1.
Dokulil, M.T., Jagsch, A., George, G.D., Anneville, O., Jankowski, T., Wahl, B., Lenhart, B., Blenckner, T.and Teubner, K. 2006. Twenty years of spatially coherent deepwater warming in lakes across Europe related to the North Atlantic Oscillation. Limnology and Oceanography 51: 2787-2793.
Doyle, M.W., Stanley, E.H., Orr, C.H., Selle, A.R., Sethi, S.A. and Harbor, J.M. 2005. Stream ecosystem response to small dam removal: Lessons from the Heartland. Geomorphology 71: 227–244.
Durance, I., and Ormerod, S.J. 2007. Climate change effects on upland stream macroinvertebrates over a 25-year period. Global Change Biology 13: 942–957.
Duarte, C. M., Conley, D. J., Carstensen, J. and Sánchez-Camacho, M. 2009. Return to Neverland: shifting baselines affect eutrophication restoration targets. Estuaries and Coasts. 32: 29–36.
EEA. 2004. Impacts of Europe's changing climate: An indicator-based assessment.EEA report no. 2/2004. European Environment Agency, Copenhagen.

EEA. 2007. Halting the loss of biodiversity by 2010: proposal for a first set of indicators to monitor progress in Europe. EEA technical report 11/2007, 38 pp. Office for Official Publications of the European Communities, Luxembourg.
Ehrman, T.P. and Lamberti, G.A. 1992. Hydraulic and particulate matter retentioni n a 3rd-order Indiana stream. J. N. Am. Benthol. Soc. 11: 341-349.
Elliott, M. Burdon, D., Hemingway, K.L. and Apitz, S.E. 2007. Estuarine, coastal and marine ecosystems restoration:Confusing management and science - a revision of concepts. Estuarine Coastal and Shelf Science 74: 349-366.
Evans, C. D., J. M. Cullen, C. Alewell, J. Kopacek, A.Marchetto, F. Moldan, A. Prechtel, M. Rogora, J. Vesely, and R. Wright. 2001. Recovery from acidification in European surface waters. Hydrology and Earth System Sciences 5: 283–297.
Evans, C.D., Monteith, D.T.and Harriman, R. 2001. Long-term variability in the deposition of marine ions at west coast sites in the UK Acid Waters Monitoring Network: impacts on surface water chemistry and significance for trend determination. Science of the Total Environment 265: 115-129.
Evans, C., Reynolds, B., Hinton, C., Hughes, S., Norris, D., Grant, G., and Williams, B.: Effects of decreasing acid deposition and climate change on acid extremes in an upland stream, Hydrol. Earth Syst. Sci., 12: 337-351
Faulkenham, S.E., Hall, R.I., Dillon, P.J. and Karst-Riddoch, T. 2003. Effects of drought-induced acidification on diatom communities in acid-sensitive Ontario lakes Limonology and Oceanography 48: 1662-1673
Fausch, K.D., Torgersen, C.E., Baxter, C.V. and Li, H.W. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52: 483-498.
Feely, R.A., Sabine, C.L., Lee, K., Berelson, W., Kleypas, J., Fabry, V.J. et al. 2004. Impact of anthropogenic CO2 on the CaCO3 system in the oceans. Science 305: 362–366.
Feld, C.K., Birk, S., Bradley, D.C., Hering, D., Kail, J., Marzin, A., Melcher, A., Nemitz, D., Pedersen, M.L., Pletterbauer, F., Pont, D., Verdonschot, P.F.M. and Friberg, N. 2011. From natural to degraded rivers and back again: a test of restoration ecology theory and practice. In: Woodward, G. (ed.): Ecosystems in a human-modified landscape: a European perspective . Advances in Ecological Research 44: 119-209.
Feld, C.K., Hering,D., Jähnig, S., Lorenz, A.W., Rolauffs, P., Kail, J., Henter, H.-P. and Koenzen, U. 2006. Ökologische Fließgewässerrenaturierung – Erfahrungen zur Durchführung und Erfolgskontrolle von Renaturierungsmaßnahmen zur Verbesserung des ökologischen Zustands. Gutachten im Auftrag des Umweltbundesamtes, 123 pp. + Anhang
Findlay, S., Quinn, J.M., Hickey, C.W., Burrell, G. and Downes, M. 2001. Effects of land use and riparian flowpath on delivery of dissolved organic carbon to streams. Limnol. Oceanogr 46: 345-355.
Gasith, A. & Resh, V.H. (1999) Streams in Mediterranean climate regions: abiotic influences and biotic responses to predictable seasonal events. Annual Review of Ecology and Systematics 30: 51-81.
George, D.G. 2000. The impact of regional-scale changes in the weather on the long-term dynamics of Eudiaptomus and Daphnia in Esthwaite Water, Cumbria. Freshwater Biology 45: 111-121.
Gerten, D. and Adrian, R. 2000. Climate-driven changes in spring plankton dynamics and the sensitivity of shallow polymictic lakes to the North Atlantic Oscillation. Limnology and Oceanography 45: 1058-1066.
Gerten, D. and Adrian, R. 2001. Differences in the persistency of the North Atlantic Oscillation signal among lakes. Limnology and Oceanography 46: 448-455.

González-Oreja, J.A. and Sáiz Salinas, J.I. 2003. Recovery simulations of grossly polluted sediments in the Bilbao estuary. Mar Pollut Bull 46: 42–48.
Gorostiaga, J.M., Borja, A., Díez, I., Francés, G., Pagola-Carte, S., Sáiz Salinas, J.I. 2004. Recovery of benthic communities, in polluted systems. In: Borja, A. and Collins, M. (eds.): Oceanography and marine environment of the Basque Country. Elsevier Oceanogr Ser 70: 549–578.
Gregory, S., Li, H. and Li, J. 2002. The conceptual basis for ecological responses to dam removal. BioScience 52, 713–723.
Gregory, S.V., Swanson, F.J., McKee, W.A. and Cummins, K.W. 1991. An ecosystem perspective of riparian zones: focus on links between land and water. BioScience 41: 540-551.
Gulati, R. D. and Van Donk, E. 2002. Lakes in the Netherlands, their origin, eutrophication and restoration: state-of-the-art review. Hydrobiologia 478: 73-106.
Gulati, R.D., Lammens, E.H.R.R., Meijer, M.-L. andVan Donk, E. (eds). 1990. Biomanipulation – Tool forWater Management. Developments in Hydrobiology 61. Kluwer Academic Publishers, Dordrecht: 628 pp. Reprinted from Hydrobiologia 200/201.
Gulati, R.D., Pires, L.M.D. and Van Donk, E. 2008. Lake restoration studies: Failures, bottlenecks and prospects of new ecotechnological measures. Limnologica 38: 233–247.
Gurnell, A.M., Gregory, K.J., Petts, G.E. 1995. The role of coarse woody debris in forest aquatic habitats: implications for management. Aquat. Conserv. 5: 143-166.
Haase, P., Pauls, S.U., Engelhardt, C.H.M. and Sundermann, A. Effects of sampling microhabitats with low coverage within the STAR/AQEM macroinvertebrate sampling protocol on stream assessment. Limnologica 38: 14-22.
Hancock, P.J. 2002. Human impacts on the stream-groundwater exchange zone. Environ. Manag. 29: 763-781.
Hansen, H.O., Boon, P.J., Madsen, B.L. & Iversen, T.M. (1998) River restoration. The physical dimension. A series of papers presented at the International Conference River Restoration '96, organized by the European Centre for River Restoration, Silkeborg, Denmark. Aquatic Conservation: Marine and Freshwater Ecosystems. 8 (1): 1-264.
Hansson, L.A., Annadotter, H., Bergmann, E., Hamrin, S.F., Jeppesen, E., Kairesalo, T., Luokkanen, E., Nilsson, P.A., Søndergaard, M. and Strand, J. 1998. Biomanipulation as an application of foodchain theory: constraints, synthesis, and recommendations for temperate lakes. Ecosystems: 1: 558–574.
Harley, C.D.G., Hughes, A.R., Hultgren, K.M., Miner, B.G., Sorte, C.B.J., Thornber, C.S., Rodriguez, L.F., Tomanek, L. and Williams, S.L.2006. The impacts of climate change in coastal marine systems. Ecology Letters 9(2): 228-241.
Hart, D.D., Johnson, T.E., Bushaw-Newton, K.L., Horwitz, R.J., Bednarek, A.T., Charles, D.F., Kreeger, D.A. and Velinsky, D.J. 2002. Dam removal: Challenges and opportunities for ecological research and river restoration. BioScience 52: 669–681.
Havel, J. and Medley, K. 2006. Biological invasions across spatial scales: intercontinental, regional, and local dispersal of Cladoceran zooplankton. Biological Invasions 8: 459-473.
Havens, K.E. 1993. Pelagic food-web structure in acidic adirondack mountain, new-york, lakes of varying humic content. Canadian Journal of Fisheries and Aquatic Sciences 50: 2688-2691.
Havens, K.E., Jin, K.R., Rodusky, A.J., Sharfstein, B., Brady, M.A., East, T.L., Iricanin, N., James, R.T., Harwell, M.C.and Steinman, A.D. 2001. Hurricane effects on a shallow lake ecosystem and its response to a controlled manipulation of water level. Scientific World Journal 1: 44-70.
Hecky, R.E., Smith, R.E.H., Barton, D.R., Guildford, S.J., Taylor, W.D., Charlton, M.N.and T. Howell. 2004. The nearshore phosphorus shunt: a consequence of ecosystem engineering by

dreissenids in the Laurentian Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 61: 1285-1293.

Henley, W.F., Patterson M.A., Neves R.J. and Lemly, AD. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. Rev. Fish. Sci. 8: 125-139.

Henriksen, L. and Brodin, Y.W. 1995. Liming of acidified surface waters: a Swedish synthesis. 458 p, Springer.

Higgins, S.N. and Vander Zanden, J. 2010. What a difference a species makes: a meta-analysis of dreissenid mussel impact on freshwater ecosystems. Ecological Monographs. 80: 179-196.

Hildrew, A. G., andOrmerod, S. J. 1995. Acidification-causes, consequences and solutions. Pages 147–160, in D. M. Harper, A. J. D. Ferguson, and R. W. Edwards (editors). The ecological basis for river management.Wiley, Chichester, UK.

Hosper, S.H., Meijer, M.-L., Gulati, R.D. and Van Donk, E. 2005. Biomanipulation in shallow lakes: concepts, case studies and perspectives. In: O'Sullivan, P.E., Reynolds, C.S. (Eds.): The Lakes Handbook, Vol. 2. Lake Restoration and Rehabilitation. Blackwell, Malden, MA, pp. 462–482.

Hughes, T.P., Baird, A.H., Bellwood, D.R., Card, M., Connolly, S.R., Folke, C. et al. 2003. Climate change, human impacts, and the resilience of coral reefs. Science 301: 929–933.

Hynes, H.B.N. 1975. The stream and its valley. Verh. Int. Ver. Theor. Ang. Limnol. 19: 1-15.

Irfanullah, H. M. and Moss, B. 2005. Effects of pH and predation by Chaoborus larvae on the plankton of a shallow and acidic forest lake. Freshwater Biology 50: 1913-1926.

Jähnig, S.C., Brabec, K., Buffagni, A., Erba, S., Lorenz, A.W., Ofenböck, T., Verdonschot, P.F.M. and Hering, D. 2010. A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. Journal of Applied Ecology 47: 671–680.

Jansson, R., Nilsson, C. and Malmqvist, B. 2007. Restoring freshwater ecosystems in riverine landscapes: the roles of connectivity and recovery processes. Freshw. Biol. 52: 589-596.

Jensen, J.P., Jeppesen, E., Kristensen, P., Bondo, P., Sondergaar, C.and Søndergaard, M. 1992. Nitrogen loss and denitrification as studied in relation to reductions in nitrogen loading in a shallow, hypertrophic lake (Lake Sobygard, Denmark). Internationale Revue der Gesamten Hydrobiologie 77: 29-42.

Jeppesen, E., Jensen, J.P., Kristensen, P., Søndergaard, M.,Mortensen, E., Sortkjaer, O., Olrik, K. 1990. Fish manipulation as a lake manipulation tool in shallow, eutrophic, temperate lakes 2: treshold levels, long-term stability and conclusions. Hydrobiologia 200/201: 219–227.

Jeppesen, E., Jensen, J.P., Kristensen, P., Søndergaard, M., Mortensen, E., Sortkjær, O. and Olrik, K. 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic, temperate lakes. II. Threshold levels, long-term stability and conclusions. Hydrobiologia 200/201: 219–227.

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K.E., Anneville, O., Carvalho, L., Coveney, M.F., Deneke, R., Dokulil, M.T., Foy, B., Gerdeaux, D., Hampton, S.E., Hilt, S., Kangur, K., Kohler, J., Lammens, E., Lauridsen, T.L., Manca, M., Miracle, M.R., Moss, B., Noges, P., Persson, G., Phillips, G., Portielje, R., Schelske, C.L., Straile, D., Tatrai, I., Willen, E.and Winder, M. 2005. Lake responses to reduced nutrient loading - an analysis of contemporary long-term data from 35 case studies. Freshwater Biology 50: 1747-1771.

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K.E., Anneville, O., Carvalho, L., Coveney, M.F., Deneke, R., Dokulil, M.T., Foy, B., Gerdeaux, D., Hampton, S.E., Hilt, S., Kangur, K., Kohler, J., Lammens, E., Lauridsen, T.L., Manca, M., Miracle, M.R., Moss, B., Noges, P., Persson, G., Phillips, G., Portielje, R., Schelske, C.L., Straile, D., Tatrai, I., Willen, E.and Winder, M., 2005. Lake responses to reduced nutrient loading - an analysis of contemporary long-term data from 35 case studies. Freshwater Biology 50: 1747-1771.

Johnson, R. K., S. Bell, L. Davies, S. Declerck, E. Laczko, M. Tonder, and Y. Uzunov. 2003. Wetlands. Pages 98–122 inJ. Young, P. Nowicki, D. Alard, K. Henle, R. K. Johnson, S. Matouch, J. Niemela¨ , and A. Watt (editors). Conflicts between human activities and the conservation of biodiversity in agricultural landscapes, grasslands, forests, wetlands, and in Europe. A report of the BIOFORUM project. Centre for Ecology and Hydrology, Banchory, Scotland.

Johnson, R.K.. and Angeler D.G. 2010. Tracing recovery under changing climate: response of phytoplankton and invertebrate assemblages to decreased acidification. Journal of the North American Benthological Society, 29(4): 1472-1490

Johnson, R. K. and D. Hering. 2009. Response taxonomic groups in streams to gradients in resource and habitat characteristics. Journal of Applied Ecology 46: 175–186.

Johnson, R. K., D. Hering, M. T. Furse, and Verdonschot, P.F.M. 2006b. Indicators of ecological change:comparison of the early response of four organismgroups to stress gradients. Hydrobiologia 566: 139–152.

Jones, P.D., Jonsson, T. and Wheeler, D. 1997. Extension to the North Atlantic Oscillation using early instrumental pressure observations from Gibraltar and South-West Iceland. International Journal of Climatology 17: 1433-1450.

Jones, P.D., Osborn, T.J. and Briffa, K.R. 2003. Pressure-based measures of the North Atlantic Oscillation (NAO): a comparison and an assessment of changes in the strength of the NAO and its influence on surface climate parameters. Geophysical Monographs 134: 51:62.

Jowett, I.G., Richardson, J. and Boubée, J.A. 2009. Effects of riparian manipulation on stream communities in small streams: Two case studies. N. Z. J. Mar. Freshwat. Res. 43: 763–774.

Jurado-Molina, J. and Livingston, P. 2002. Climate-forcing effects on trophically linked groundfish populations: implications for fisheries management. Can. J. Fish. Aquat. Sci. 59: 1941–1951.

Kail, J., Hering, D., Muhar, S., Gerhard, M. and Preis, S. (2007) The use of large wood in stream restoration: experiences from 50 projects in Germany and Austria. J. Appl. Ecol. 44: 1145–1155.

Kangur, A., Kangur, P. and Pihu, E. 2002. Long-term trends in the fish communities of Lakes Peipsi and Vortsjarv (Estonia). Aquatic Ecosystem Health & Management 5: 379-389.

Keizer-Vlek, H.E., Michels, H. and Verdonschot, P.F.M. 2011. Case study Vecht catchment – Changes in ecological condition of streams in relation to land use change and hydromorphological restoration measures. In : Feld, C.K. (ed.) Deliverable D5.1-2: Driver-Pressure-Impact and Response-Recovery chains in European rivers: observed and predicted effects on BQEs.

Kleypas, J.A., Buddemeier, R.W., Archer, D., Gattuso, J.-P., Langdon, C. and Opdyke, B.N. 1999. Geochemical consequences of increased atmospheric carbon dioxide on coral reefs. Science 284: 118–120.

Kolpin, D.W., Furlong, E.T., Meyer, M.T., Thurman, E.M., Zaugg, S.D., et al. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in US streams, 1999-2000: a national reconnaissance. Environ. Sci. Technol. 36: 1202-1211.

Kondolf, G.M., Boulton, A.J., O'Daniel, S., Poole, G.C., Rahel, F.J., Stanley, E.H., Wohl, E., Bång, A., Carlstrom, J., Cristoni, C., Huber, H., Koljonen, S., Louhi, P. and Nakamura, K. 2006. Process-based ecological river restoration: visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages. Ecology and Society 11(2): 5.

Kristensen, P. & Hansen, H.O. (eds.) (1994) European Rivers and Lakes: Assessment of their Environmental State. European Environment Agency Environmental Monographs 1, Copenhagen.

Lake P. S. 2001. Restoring streams: re-building and reconnecting. In: Brisbane. Rutherfurd, I., Sheldon, F., Brierley, G. and Kenyon, C. (eds): Third Australian Stream Management Conference. Cooperative Research Centre for Catchment Hydrology, Canberra. pp. 369–371.
Lake, P.S., Bond, N. and Reich, P. 2007. Linking ecological theory with stream restoration. Freshw. Biol. 52: 597-615.
Lambin, E.F., Baulies, X., Bockstael, N., Fischer, G., Krug, T., Leemans, R., Moran, E.F., Rindfuss, R.R., Sato, Y., Skole, D., Turner, B.L. II, Vogel, C., 1999. Land-use and land-cover change (LUCC): Implementation strategy. IGBP Report No. 48, IHDP Report No. 10, Stockholm, Bonn.
Lammens, E.H.R.R., Gulati, R.D., Meijer, M.L. & Van Donk, E. 1990. The first biomanipulation conference: a synthesis. Hydrobiologia 200/201: 619–627.
Larson, M.G., Booth, D.B. and Morley, S.A. 2001. Effectiveness of large woody debris in stream rehabilitation projects in urban basins. Ecol. Eng. 18: 211–226.
Larssen, B. 1995. The effects of liming on aquatic floar. Pp 193-261, In: Liming of acidified surface waters: A Swedish synthesis (Henriksen and Brodin, eds). Springer.
Lenat, D.R. and Crawford, J.K. 1994. Effects of landuse on water-quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia 294: 185-199.
Levell, A. P. and Chang, H. 2008. Monitoring the channel process of a stream restoration project in an urbanizing watershed: A case study of Kelley Creek, Oregon, USA. River Res. Appl. 182: 169–182.
Lewison, R.L., Freeman, S.A. and Crowder, L.B. 2004.Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. Ecol. Lett. 7: 221–231.
Liess, M. and Schulz, R. 1999. Linking insecticide contamination and population response in an agricultural stream. Environ. Toxicol. Chem. 18: 1948-1955.
Lintermans, M. 2004. Human-assisted dispersal of alien freshwater fish in Australia. New Zealand Journal of Marine and Freshwater Research 38: 481–501.
Livingstone, D. M., 1999. Ice break-up on southern Lake Baikal and its relationship to local and regional air temperatures in Siberia and to the North Atlantic Oscillation. Limnology and Oceanography 44: 1486-1497.
Livingstone, D.M. 2000. Large scale climatic forcing detected in historical observations of ice break-up. Verhandlungen IVL 27: 2775-2783.
Livingstone, D.M. and Dokulil, M.T. 2001. Eighty years of spatially coherent Austrian lake surface temperatures and their relationship to regional air temperature and the North Atlantic Oscillation. Limnology and Oceanography 46: 1220-1227.
Lotze, H.K., Coll, M., Magera, A.M., Ward-Paige, C. and Airoldi, L. 2011. Recovery of marine animal populations and ecosystems. Trends in Ecology and Evolution 26(11): 595-605.
Louhi, P., Mykra, H., Paavola, R., Huusko, A., Vehanen, T., Mäki-Petäys, A. and Muotka, T. 2011. Twenty years of stream restoration in Finland: little response by benthic macroinvertebrate communities. Ecological Applications 21(6): 1950–1961.
Lowrance, R., Todd, R., Fail, J.J., Hendrickson, O.J., Leonard, R., Asmussen, L. 1984. Riparian forests as nutrient filters in agricultural watersheds. BioScience 34: 374-377.
Macdonald, D.W., Moorhous, T.P. and Enkc, J.W. 2002. The ecological context: a species population perspective. In: Perrow, M.R. and Davy, A.J. (eds.): Handbook of Ecological Restoration. Volume 1. Cambridge University Press, NY, USA.
Mainstone, C.P. and Parr, W. 2002. Phosphorus in rivers--ecology and management. Sci. Total Environ. 282: 25-47.
Manchester, S. J. and Bullock, J.M. 2000. The impacts of non-native species on UK biodiversity and the effectiveness of control. Journal of Applied Ecology 37: 845-864.

Mander, L., Cutts, N.D., Allen, J. and Mazik, K. 2007. Assessing the development of newly created habitat for wintering estuarine birds. Estuarine, Coastal and Shelf Science 75: 163–174.

Maridet, L., Wasson, J.G., Philippe, M. and Amoros, C. 1995. Benthic organic-matter dynamics in three streams-riparian vegetation or bed morphology control. Arch. Hydrobiol. 132: 415-425.

Martin, T.L., Kaushik, N.K., Trevors, J.T., Whiteley, H.R. 1999. Review: denitrification in temperate climate riparian zones. Water Air Soil Poll. 111: 171-186.

Matsuzaki, S., Usio, N., Takamura, N.and Washitani, I. 2009. Contrasting impacts of invasive engineers on freshwater ecosystems: an experiment and meta-analysis. Oecologia. 158: 673-686.

Matthews, J., Reeze, B., Feld, C.K. and Hendriks, A.J. 2010. Lessons from practice: assessing early progress and success in river rehabilitation. Hydrobiologia 655: 1-14.

May, L. and Carvalho, L. 2010. Maximum growing depth of macrophytes in Loch Leven, Scotland, United Kingdom, in relation to historical changes in estimated phosphorus loading. Hydrobiologia 646: 123-131.

Mazik, K., Smith, J.E., Leighton, A. and Elliott, M.. 2007. Physical and biological development of a newly breached managed realignment site, Humber estuary, UK. Marine Pollution Bulletin 55: 564–578.

McKee, D., Atkinson, D., Collings, S.E., Eaton, J.W., Gill, A.B., Harvey, I., Hatton, K., Heyes, T., Wilson, D. and Moss, B. 2003. Response of freshwater microcosm communities to nutrients, fish and elevated temperature during winter and summer. Limnology and Oceanography 48: 707–722.

Mehner, T., Benndorf, J., Kasprzak, P. and Koschel, R. 2002. Biomanipulation of lake ecosystems: successful applications and expanding complexity in the underlying science. Freshwater Biology 47: 2453–2465.

Meijer, M.L., de Bois, I., Scheffer, M., Portielje, R. and Hosper, H. 1999. Biomanipulation in shallow lakes in The Netherlands an evaluation of 18 case studies. Hydrobiologia 408/409: 13–30.

Meyer, J.L., Sale, M.J., Mulholland, P.J. and Poff, N.L. 1999. Impacts of climate change on aquatic ecosystem functioning and health. J. Am. Water Resour Assoc. 35: 1373-1386.

Meyer, W.B. and Turner, B.L., eds. 1994. Changes in Land Use and Land Cover: A Global Perspective. New York: Cambridge Univ. Press. 537 pp.

Miller, S.W., Budy, P. and Schmidt, J.C. 2010. Quantifying macroinvertebrate responses to in-stream habitat restoration: applications of meta-analysis to river restoration. Restoration Ecology 18: 8–19.

Mills, E.L., Leach, J.H., Carlton, J.T. and Secor, C.L. 1994.Exotic Species and the Integrity of the Great Lakes. BioScience 44(10): 666-676.

Monteith, D.T., Evans, C.D. and Reynolds, B. 2000. Are temporal variations in the nitrate content of UK upland freshwaters linked to the North Atlantic Oscillation? Hydrological Processes 14: 1745-1749.

Moss, B., McKee, D., Atkinson, S., Collings, S.E., Eaton, J.W., Gill, A.B., Harvey, I., Hatton, K., Heyes, T. and Wilson, D. 2003 How important is climate? Effects of warming, nutrient addition and fish on phytoplankton in shallow lake microcosms. Journal of Applied Ecology 40: 782–792.

Nilsson, C. and Berggren, K. 2000. Alterations of riparian ecosystems caused by river regulation. BioScience 50: 783-792.

Niyogi, D.K., Simon, K.S. and Townsend C.R. 2003. Breakdown of tussock grass in streams along a gradient of agricultural development in New Zealand. Freshw. Biol. 48: 1698-1708.

Noges, P., Mischke, U., Laugaste, R. and Solimini, A.G. 2010. Analysis of changes over 44 years in the phytoplankton of Lake Vrtsjarv (Estonia): the effect of nutrients, climate and the investigator on phytoplankton-based water quality indices. Hydrobiologia 646: 33-48.
Økland, J., and Økland, K.A.. 1986. The effects of acid deposition on benthic animals in lakes and streams. Experentia 42: 471–486.
Ormerod, S.J. & Durance, I. 2009. Restoration and recovery from acidification upland Welsh streams over 25 years. Journal of Applied Ecology, 46: 164–174.
Orr, C.H., Rogers, K.L. and Stanley, E.H. 2006. Channel morphology and P uptake following removal of a small dam. J. North Am. Benthol. Soc. 25: 556–568.
Orth, R.J., Williams, M.R., Marion, S.R., Wilcox, D.J., Carruthers, T.J.B., Moore, K.A., Kemp, W.M., Dennison, W.C., Rybicki, N., Bergstrom, P. and Batiuk, R.A. 2010. Long-term trends in submersed aquatic vegetation (SAV) in Chesapeake Bay, USA, related to water quality. Estuaries and Coasts 33(5): 1144–1163.
Osborne, L.L. and Kovacic, D.A. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshw. Biol. 29: 243-258.
Ottersen, G., B. Planque, A. Belgrano, E. Post, P. C. Reid and Stenseth, N.C. 2001. Ecological effects of the North Atlantic Oscillation. Oecologia (Berlin) 128: 1–14.
Paillex, A., Doledec, S., Castella, E. and Merigoux, S. 2009. Large river floodplain restoration: predicting species richness and trait responses to the restoration of hydrologic connectivity. J. Appl. Ecol. 46: 250-258.
Palmer, M., Allan, J.D., Meyer, J. and Bernhardt, E.S. 2007. Restoration in the twenty-first century: data and experimental knowledge to inform future efforts. Restoration Ecology 15(3): 472–481.
Palmer, M.A. 2009. Invited Odum essay: Reforming watershed restoration: science in need of application and applications in need of science. Estuaries and Coasts 32: 1–17.
Palmer, M.A., Bernhardt, E.S., Allan, J.D., Lake, P.S., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Follstad Shah, J., Galat, D.L., Loss, S.G., Goodwin, P., Hart, D.D., Hassett, B., Jenkinson, R., Kondolf, F.M., Lave, R., Meyer, J.L., O'Donnel, T.K., Pagano, L. and Sudduth, E. 2005. Standards for ecologically successful river restoration. J. Appl. Ecol. 42: 208–217.
Palmer, M.A., Menninger, H.L. and Bernhardt, E.S., 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater Biology 55: 1–18.
Parkyn, S., Davies-Colley, R., Cooper, A. and Stroud, M. 2005. Predictions of stream nutrient and sediment yield changes following restoration of forested riparian buffers. Ecol. Eng. 24: 551–558.
Paul, M.J. and Meyer J.L. 2001. Streams in the urban landscape. Annu. Rev. Ecol. Syst. 32: 333-365.
Pearl, H. W. 2009. Controlling eutrophication along the freshwater-marine continuum: dual nutrient (N and P) reductions are essential. Estuaries and Coasts 32: 593-601.
Perrow, M.R. and Davy, A.J. (eds.) .2002. Handbook of ecological Restoration. Volume 1: Principles of restoration. Cambridge University Press, NY, USA.
Peterson, G. 2002. Contagious disturbance, ecological memory, and the emergence of landscape patterns. Ecosystems, 5(4): 329–338.
Pirrone, N., Trombino, G., Cinnirella, S., Algieri, A., Bendoricchio, G. and Palmeri, L. 2005. The Driver-Pressure-State-Impact-Response (DPSIR) approach for integrated catchment-coastal zone management: preliminary application to the Po catchment-Adriatic Sea coastal zone system. Reg Environ Change 5: 111–137.
Poff, N.L. and Allan, D.J. 1995. Functional organization of stream fish assemblages in relation to hydrologic variability. Ecology 76: 606-627.

Poff, N.L., Allan, J.D., Bain, M.B., et al. (1997) The natural flow regime: a paradigm for river conservation and restoration. Bioscience 47: 769-784.

Pollard, D.A. and Hannan, J.C. 1994. The ecological effects of structural flood mitigation works on fish habitats and fish communities in the lower Clarence River system of south-eastern Australia. Estuaries 17: 427–461.

Porto, L.M., McLaughlin, R.L. and Noakes, D.L.G. 1999. Low-head barrier dams restrict the movements of fishes in two Lake Ontario streams. North American Journal of Fisheries Management 19: 1028–1036.

Pretty, J.L., Harrison, S.S.C., Shepherd, D.J., Smith, C., Hildrew, A.G. and Hey, R.D. 2003. River rehabilitation and fish populations: assessing the benefit of instream structures. J. Appl. Ecol. 40 (2): 251–265.

Psenner, R. & Schmidt, R. 1992 Climate-driven pH control of remote alpine lakes and effects of acid deposition. Nature 356: 781−783

Quinn, J.M. 2000. Effects of pastoral development. In New Zealand Stream Invertebrates: Ecology and Implications for Management, ed. KJ Collier,M J Winterbournp, p. 208-29. ChristchurchN, Z: Caxton.

Räisänen J., U. Hansson, A. Ullerstig, R. Döscher, L. P. Graham, C. Jones, M. Meier, P. Samuelsson & U.Willén, 2003. GCM driven simulations of recent and future climate with the Rossby Centre coupled atmosphere – Baltic Sea regional climate model RCAO. SWECLIM programme and EU PRUDENCE project, report number: RMK No. 101, January 2003, 61 pp.

Reitberger, B., Matthews, J. , Feld, C.K., Davis, W. and Palmer, M.A. 2010. River monitoring and indication of restoration success: comparison of EU and BCG frameworks. University of Duisburg-Essen, Essen, 154 pp.

Resh, V. H., and D. M. Rosenberg. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, London, UK.

Reynolds, C.S., Jaworski G.H.M., Roscoe, J.V., Hewitt, D.P.and George, D.G. 1998. Responses of the phytoplankton to a deliberate attempt to raise the trophic status of an acidic, oligotrophic mountain lake. Hydrobiologia 370: 127-131.

Riel M.C. van, Velde G. van der, Rajagopal S., Marguillier S., Dehairs F.& Vaate A. bij de, 2006. Trophic relationships in the Rhine food web during invasion and after establishment of the Ponto-Caspian invader Dikerogammarus villosus. Hydrobiologia (2006) 565: 39–58.

Rip, W. 2007. Cyclic state shift in a restored shallow lake. Wageningen University, Wageningen: 216.

Rodriguez , L.F. 2006. Can invasive species facilitate native species? Evidence of how, when, and why these impacts occur. Biological Invasions 8: 927–939.

Rolland, R.M. 2000. A review of chemically induced alterations in thyroid and vitamin A status from field studies of wildlife and fish. J. Wildl. Dis. 36: 615-635.

Roni, P, Hanson, K., Beechi, T., Pess, G., Pollock, M. and Bartley, D.M. 2005. Habitat rehabilitation for inland fisheries: Global review of effectiveness and guidance for rehabilitation of freshwater ecosystems. FAO Fisheries technical paper 484. Food and Agriculture Organization of the United Nations, Rome.

Roni, P. and Quinn, T.P. 2001. Density and size of juvenile salmonids in response to placement of large woody debris in western Oregon and Washington streams. Can. J. Fish. Aquat. Sci. 58: 282–292.

Roni, P. and Quinn, T.P. 2001. Effects of Wood Placement on Movements of Trout and Juvenile Coho Salmon in Natural and Artificial Stream Channels. Trans. Amer. Fish. Soc.130: 675–685.

Roni, P., Hanson, K. and Beechie, T. (2008) Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. North Am. J. Fish. Manage. 28: 856–890.
Rosenthal, S.K., Stevens, S.S.and Lodge, D.M. 2006. Whole-lake effects of invasive crayfish (Orconectes spp.) and the potential for restoration. Canadian Journal of Fisheries and Aquatic Sciences 63: 1276-1285.
Rusak, J. A., N. D. Yan, and Somers, K.M. 2008. Regional climatic drivers of synchronous zooplankton dynamics in north-temperate lakes. Canadian Journal of Fisheries and Aquatic Sciences 65: 878–889.
Sáiz-Salinas, J.I. and González-Oreja, J.A. 2000. Stress in estuarinecommunities: lessons from the highly-impacted Bilbaoestuary (Spain). J Aquat Ecosyst Stress Recov 7: 43–55.
Sas, H., 1989. Lake restoration by reduction of nutrient loading: expectations, experiences, extrapolations. Academic Verlag Richards, Germany.
Scheffer, M. 1998. Ecology of shallow lakes. Chapman & Hall.
Scheffer, M., Straile, D., Van Nes, E.H.and Hosper, H. 2001. Climatic warming causes regime shifts in lake food webs. Limnology and Oceanography 46: 1780-1783.
Schindler, D. W. 2006. Recent advances in the understanding and management of eutrophication. Limnology and Oceanography 51: 356-363.
Schindler, D.W., Hecky, R.E., Findlay, D.L., Stainton, M.P., Parker, B.R., Paterson, M.J., Beaty, K.G., Lyng, M.and Kasian, S.E.M. 2008. Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. Proceedings of the National Academy of Sciences of the United States of America 105: 11254-11258.
Schlosser, IJ. 1991. Stream fish ecology: a landscape perspective. BioScience 41: 704-712.
Schreiber, E.S.G., Lake, P.S. and Quinn, G.P. 2002. Facilitation of native stream fauna by an invading species? Experimental investigations of the interaction of the snail, Potamopyrgus antipodarum (Hydrobiidae) with native benthic fauna. Biological Invasions 4: 317–325.
Schulz, R. and Liess, M. 1999. A field study of the effects of agriculturally derived insecticide input on stream macroinvertebrate dynamics. Aquat. Toxicol. 46:155-176.
Skjevåle, B. L., C. D. Evan, T. Larssen, A. Hindar, annd G. Raddum. 2003. Recovery from acidification in Europeansurface waters: a view to the future. Ambio 32: 170–175.
Scott, M.C. and Helfman, G.S. 2001. Native invasions, homogenization, and the mismeasure of integrity of fish assemblages. Fisheries 26: 6-15.
Shields, F.D. Jr, Copeland, R.R., Klingeman, P.C., Doyle, M.W., Simon, A. 2003. Design for stream restoration. J Hydraul Eng 129: 575–584.
Shields, F.D., Knight, S.S. and Stofleth, J.M. 2006. Large wood addition for aquatic habitat rehabilitation inan incised, sand-bed stream, Little Topashaw Creek, Mississippi. River Res. Appl. 22: 803–817.
Shurin, J.B., Cottenie, K. and Hillebrand, H. 2009. Spatial autocorrelation and dispersal limitation in freshwater organisms. Oecologia 159: 151-159.
Simenstad, C., Reed, D., and Ford, M. 2006. When is restoration not?: Incorporating landscape-scale processes to restore self-sustainingecosystems in coastal wetland restoration. Ecological Engineering 26: 27–39.
Slonosky, V.C., Jones, P.D.and Davies, T.D. 2000. Variability of the surface atmospheric circulation over Europe, 1774-1995. International Journal of Climatology 20: 1875-1897.
Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller, 2007: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Sommaruga-Wograth, S., Koinig, K.A., Schmidt, R., Sommaruga, R., Tessardi, R. and Psenner, R., 1997. Temperature effects on the acidity of remote alpine lakes. Nature 387: 64-67.

Søndergaard, M. 2007. Nutrient dynamics in lakes – with emphasis on phosphorus, sediment and lake restorations. Doctor's dissertation (DSc). National Environmental Research Institute, University of Aarhus, Denmark. 276 pp.

Søndergaard, M., Jeppesen, E., Lauridsen,T.L., Skov, C., Van Nes, E.H., Roijackers, R., Lammens, E. and Portielje, R. 2007. Lake restoration: successes, failures and long-term effects. Journal of Applied Ecology 44: 1095-1105.

Spears, B, Gunn, I., Meis, S. and May, L. 2011. Analysis of cause-effect-recovery chains for lakes recovering from eutrophication. CEH-report. Contribution to Deliverable D6.4-2.

Spears, B. M. and Jones, I. 2010. The long-term (1979-2005) effects of the north Atlantic oscillation on wind induced wave mixing in Loch Leven (Scotland). Hydrobiologia 646: 49-59.

Šporka, F., H.E. Vlek, E. Bulánková & I. Krno (2006) Influence of seasonal variation on bioassessment of streams using macroinvertebrates. Hydrobiologia 566: 543-555.

Stauffer, J.C., Goldstein, R.M. and Newman, R.M. 2000. Relationship of wooded riparian zones and runoff potential to fish community composition in agricultural streams. Can. J . Fish. Aquat. Sci. 57: 307-316.

Straile, D. 2002. North Atlantic Oscillation synchronizes food-web interactions in central European lakes. Proceedings of the Royal Society B-Biological Sciences 269: 391-395.

Straile, D., Livingstone, D.M., Weyhenmeyer, G.A.and George, D.G. 2003. The response of freshwater ecosystems to climate variability associated with the North Atlantic Oscillation. Geophysical Monographs 134: 263-279.

Strayer, D.L., Beighley, R.E., Thompson, L.C., Brooks, S., Nilsson, C., et al. 2003. Effects of land cover on stream ecosystems: roles of empirical models and scaling issues. Ecosystems 6: 407-423.

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

Stoddard, J. L., D. S. Jeffries, A. Lu¨ Kewille, T. A. Clair, P. J. Dillon, C. T. Driscoll, M. Forsius, M. Johannessen, J. S. Kahl, J. H. Kellogg, A. Kemp, J. Mannio, D. T. Monteith, P. S. Murdoch, S. Patrick, A. Rebsdorf, B. L. Skjeklvåle, M. P. Stainton, T. Traaen, H. van Dam, K. E. Webster, J. Wieting, and Wilander, A. 1999. Regional trends in aquatic recovery from acidification in North America and Europe. Nature 401: 575–578.

Strong, K. F., and Robinson, G. 2004. Odonate communitiesof acidic Adirondack Mountain lakes. Journal of the North American Benthological Society 23: 839–852.

Sundermann, A., Stoll, S. and Haase, P. 2011. River restoration success depends on the species pool of the immediate surroundings. Ecological Applications, 21(6): 1962–1971.

Sutherland, A.B., Meyer J.L. and Gardiner, E.P. 2002. Effects of land cover on sediment regime and fish assemblage structure in four southern Appalachian streams. Freshw. Biol. 47: 1791-1805.

Svensson, J.E., Henriksen, L, Larsson, S. and Wilander, A. Liming strategies and effects: the Lake Gårdsjön case study. Pp 309-323, In: Liming of acidified surface waters: A Swedish synthesis (Henriksen and Brodin, eds). Springer.

Swedish Environmental Protection Agency (SEPA) (2007) Economic Instruments in Environmental Policy. Report 5678. SEPA (Naturvårdsverket), Stockholm, Sweden. Available at: http://www.naturvardsverket.se/ Documents/publikationer/620-5678-6.pdf (accessed on 9 June 2009)

Thomson, J.R., Hart, D.D., Charles, D.F., Nightingale, T.L. and Winter, D.M. 2005. Effects of removal of a small dam on downstream macroinvertebrate and algal assemblages in a Pennsylvania stream. J. North Am. Benthol. Soc. 24: 192–207.

Townsend, C.R. and Hildrew, A.G. 1976. Field experiments on the drifting, colonization and continuous redistribution of stream benthos. J. Anim. Ecol. 45: 759-772.
Townsend, C.R., Doledec, S., Norris, R., Peacock, K. and Arbuckle, C. 2003. The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. Freshw. Biol. 48: 768-785.
Trannum, H.C., Olsgard, F., Skei, J.M., Indrehus, J., Overas, S. and Eriksen, J. 2004. Effects of copper, cadmium and contaminated harbour sediments on recolonisation of soft-bottom communities. J Exp Mar Biol Ecol 310: 87–114.
Turner, B.L. II, Clark, W.C., Kates, R.W., Richards, J.F., Mathews, J.T., Meyer, W.B. (Eds.), 1990. The Earth as Transformed by Human Action: Global and Regional Changes in the Biosphere Over the Past 300 Years. Cambridge Univ. Press, Cambridge.
Van de Bund, W.J. and Van Donk, E. 2002. Short-term and long-term effects of zooplanktivorous fish removal in a shallow lake: a synthesis of 15 years of data from Lake Zwemlust. Freshwater Biology, 47: 2380–2387.
Van der Wal, D., Pye, K. and Neal, A. 2002. Long-term morphological change in the Ribble Estuary, northwest England. Mar. Geol. 189 (3–4): 249–266.
Van Donk, E., Gulati, R.D., Iedema, A.and Meulemans, J.T. 1993. Macrophyte-related shifts in the nitrogen and phosphorus contents of the different trophic levels in a biomanipulated shallow lake. Hydrobiologia 251: 19-26.
Vannote, R.L., Minshall, W.G., Cummins, K.W., Sedell, J.R., Cushing, C.E. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37: 130-137.
Violin, C. R., P. Cada, E. B. Sudduth, B. A. Hassett, D. L. Penrose, and E. S. Bernhardt. 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications 21: 1932–1949.
Walser, C.A. and Bart, H.L. 1999. Influence of agriculture on in-stream habitat and fish community structure in Piedmont watersheds of the Chattahoochee River System. Ecol. Freshw. Fish 8: 237-246.
Walsh, C.J., Fletcher, T.D.and Ladson, A.R. 2005. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. Journal of the American Benthological Society 24: 690-705.
Walsh, C.J., Sharpe A.K., Breen, P.F. and Sonneman, J.A. 2001. Effects of urbanization on streams of the Melbourne region, Victoria, Australia. I. Benthic macroinvertebrate communities. Freshw. Biol. 46: 535-551.
Wang, L., Lyons, J. and Kanehl, P. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environ. Manag. 28: 255-266.
Weyhenmeyer, G. A. 2004. Synchrony in relationships between the North Atlantic Oscillation and water chemistry among Sweden’s largest lakes. Limnology and Oceanography 49: 1191–1201.
Weyhenmeyer, G.A., Jeppesen, E., Adrian, R., Arvola, L., Blenckner, T., Jankowski, T., Jennings, E., Noges, P., Noges, T. and Straile, D. 2007. Nitrate-depleted conditions on the increase in shallow northern European lakes. Limnology and Oceanography 52: 1346-1353.
Weyhenmeyer, G.A. 2004. Synchrony in relationships between the North Atlantic Oscillation and water chemistry among Sweden’s largest lakes. Limnology and Oceanography 49: 1191-1201.
Weyhenmeyer, G.A., Westoo, A.K. and Willen, E. 2008. Increasingly ice-free winters and their effects on water quality in Sweden's largest lakes. Hydrobiologia 599: 111-118.
Wiens, J.A. 2002. Riverine landscapes: taking landscape ecology into the water. Freshw. Biol. 47: 501-515.

Williams, M.R., Filoso, S., Longstaff, B.J. and Dennison, W.C. 2010. Long-Term Trends of Water Quality and Biotic Metrics in Chesapeake Bay: 1986 to 2008. Estuaries and Coasts 33: 1279–1299.

Withers, P. J. A. & P. M. Haygarth, 2007. Agriculture, phosphorus and eutrophication: a European perspective. Soil Use and Management 23: 1-4.

Wood, P.J. and Armitage, P.D. 1997. Biological effects of fine sediment in the lotic environment. Environ. Manag. 21: 203-217.

Woodward, D.F., Goldstein, J.N., Farag, A.M. and Brumbaugh, W.G. 1997. Cutthroat trout avoidance of metals and conditions characteristic of a mining waste site: Coeur d'Alene River, Idaho. Trans. Am. Fish. Soc. 126: 699-706.

Yan, N. D., B. Leung, B. Keller, S. E. Arnott, J. M. Gunn, and Raddum, G. G.. 2003. Developing conceptual frameworksfor the recovery of aquatic biota from acidification.Ambio 32: 165–169.

Yan, N. D., B. Leung, B. Keller, S. E. Arnott, J. M. Gunn, and Raddum, G. G.. 2003. Developing conceptual frameworksfor the recovery of aquatic biota from acidification.Ambio 32: 165–169.