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SEVENTH FRAMEWORK PROGRAMME

# WISER

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

# DELIVERABLE

## Deliverable D6.4-2:
## Report on the differences between cause-effect-recovery chains of different drivers within water categories

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| Dissemination Level | | |
| PU | Public | X |
| PP | Restricted to other programme participants (including the Commission Services) | |
| RE | Restricted to a group specified by the consortium (including the Commission Services) | |
| CO | Confidential, only for members of the consortium (including the Commission Services) | |

# Non-technical summary

The WFD aims to combine catchment scale understanding across a range of aquatic ecosystems to improve ecological status within specific river basins. Catchment-wide integrated river basin management requires knowledge on cause-effect-recovery chains within water bodies as well as on the interactions between water bodies and categories.

The aim of Deliverable 6.4-2 is to compare differences between cause-effect-recovery chains of different drivers/stressors within different water categories for different organism groups. To meet the deliverable's aim, the current body of literature was surveyed for recovery studies. More specific recovery processes from eutrophication and acidification in lakes and from hydromorphological degradation in rivers. For estuarine and coastal (marine) waters, different anthropogenic pressures were studied.

There is a common agreement that the drivers and pressures in general are the same in lakes, rivers estuaries 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 estuaries and coastal waters were diverse in drivers and pressures studied, although they are fairly limited.

In lakes most studies dealt with measures to reduce eutrophication, either source- or effect-related, to decrease phosphorus loads (and to a lesser extent nitrogen loads). The response of organism groups were studied within the food web relations or cascades. The lake acidification studies considered liming. Liming is an effect-related measure that has to be repeated several times to show an effect. Response was related to indicators of the reference and diversity.

Stream restoration was studied for weir and dam removal, remeandering, instream habitat enhancement and re-introduction of riparian buffers. As to be expected, weir and dam removal improved connectivity for fauna, while all other groups of measures rather focus on general habitat improvement. The response was always expressed in species composition and diversity.

Estuarine and coastal restoration projects were scarce. Studies strongly differed in type of stressor. The response was always expressed in biomass, species composition, richness and diversity.

The differences between cause-effect-recovery chains of different stressors within lakes, streams and estuarine/coastal waters for different organism groups are comparable on a generic level, but largely differ at site level. The effects on organism groups were mostly measured in comparable terms of composition and diversity, but of course strongly differ between taxa identity. Functional aspects remain mostly unconsidered and, hence, unknown.

# Report on the differences between cause-effect-recovery chains of different drivers within water categories

## 1. Introduction

The WFD aims to remove the traditional dichotomous approach to environmental management by combining catchment scale understanding across a range of aquatic ecosystems to improve ecological status within specific river basins. This requires an assessment of the ecological responses and interactions across lakes, rivers and estuaries related to eutrophication, hydromorphological change, and acidification.

In Workpackage 6.4 of the WISER project knowledge is gathered to support catchment wide integrated basin management. 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. It needs knowledge on main driver-pressor-stressor-impact-recovery chains per water category and organism group. Furthermore, it should relate to processes related to biology (connectivity, metapopulation and dispersal) and global change (climate change, land and water use). The main stressors studied in WP6.4 are listed in Table 1.

Table 1. Key stressors studied per water category in WP6.4.

| **water category** | **stressor** |
|---|---|
| lakes | eutrophication |
| | acidification |
| rivers | acidification |
| | hydromorphology - remeandering |
| | hydromorphology - |
| | hydromorphology - |
| estuarine and coastal marine waters | eutrophication/organic pollution |
| | metals |
| | habitat degradation |

The aim of Deliverable 6.4.2. is to compare differences between cause-effect-recovery chains of different drivers/stressors within different water categories for different organism groups.

Within each water category the common approach is to perform a literature survey, if possible supported by a meta-analysis, taking the following questions into account:

- What is reported on processes and functional features?
- What about over-arching biological processes and global change?
- Are there antagonistic, neutral, additive or synergistic characteristics described of the impact of multiple stressors within the respective water category?

Overall, we want to detect commonalities among different cause-effect-recovery chains to develop a method to combine recovery effects in a summarising ‘catchment’ metric that will be part of Deliverable 6.4.3.

## 2. Methods

To meet the deliverable aim the current literature, both scientific and grey literature, was surveyed for recovery studies.

To disentangle recovery processes from eutrophication in lakes a meta-analysis of over 743 lake-equivalent case studies from 364 peer reviewed publications was performed. In addition, site-specific data from lake recovery case studies have been used to examine the responses of freshwater lakes to eutrophication management in terms of their ecological structure and function.

Recovery from acidification in lakes was studied by surveying the ISI Web of Knowledge and Google Scholar using the key words “acidification”, “lakes”, “liming”.
In addition, for acidification in lakes data from the Swedish lake monitoring program to perform additional analyses for further exploring trends that were deemed worthwhile for further investigationof interest. These analyses are currently at different progress stages and will be prepared for publication in scientific journals.

The rivers literature survey was conducted using the ISI Web of Knowledge and SCOPUS. The focus was on publications in peer-reviewed journals (and references therein), which was then extended by selected peer-reviewed reports, grey literature and other publications using Google Scholar and further web search engines. Major search terms were restoration, rehabilitation in combination with riparian vegetation or buffer, large wood or large woody debris, habitat, bed or channel structure, weir or dam removal, remeandering, for fish, invertebrates, macrophytes, phytobenthos and algae in streams or rivers.

Altogether, 36 references were analysed to develop the Conceptual Model on streams – hydrology interaction by weir and dam removal (< 5 m height). Among them, 22 papers represent active weir removal case studies, another nine review the effects of weir removal, and five additional references provide basic ecological relationships between related habitat modifications and aquatic organisms in streams.

For the morphology - (stretch scale) remeandering topic in total 91 projects were examined within grey and peer reviewed literature. Relevant information was extracted and categorized within an excel sheet which is supplementary to this chapter.

The morphology – (site scale) instream mesohabitat enhancement identified 132 peer-reviewed references, 75 of which fulfilled the criteria defined for this review.

For estuarine and coastal marine waters, we used Borja et al. (2010) and the literature therein. The review surveyed 51 longterm 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. These case studies are from 23 different anthropogenic pressures and include different geographical regions (19 countries from all continents).

## 3. Results

### *3.1 Lakes - eutrophication*

<u>*Drivers and pressures*</u>

The analysis of lake recovery from eutrpophication was conducted using the Driver, Pressure, State, Impact, Response (DPSIR) framework modified to include the recovery phase (Figure 1).

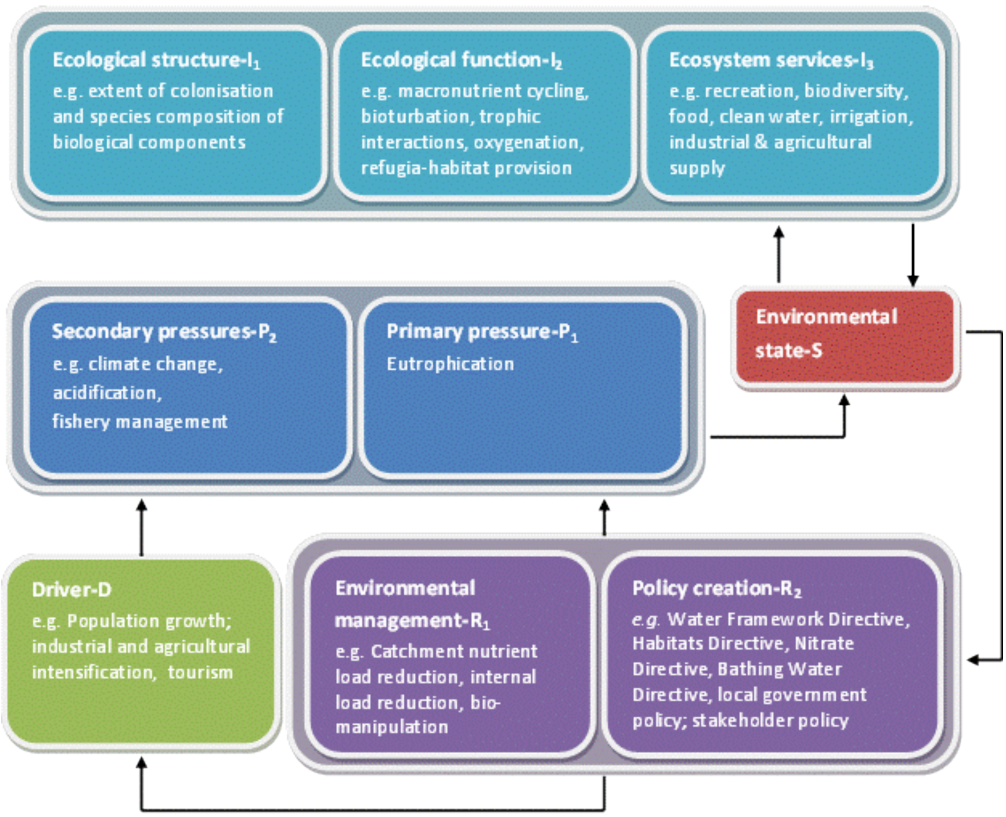



*Figure 1. Conceptual diagram of the Driver, Pressure, State, Impact and Response (DPSIR) framework as it applies to the management of eutrophic lakes.*

The main drivers of eutrophication identified in the literature included population growth, industrialisation, agricultural intensification, tourism and recreation. These drivers were associated with a wide range of primary and secondary pressures (Table 2). The key primary eutrophication pressures were related to agricultural sources (e.g. animal waste, fertiliser applications, soil erosion), and discharges from industry (e.g. aquaculture, paper mills, food manufacturing) and the infrastructure associated with areas of high population density (e.g. waste water treatment works, housing, roads). Acidification and fishery management were the most commonly reported secondary pressures (29% and 20%, respectively), followed by industrial pollution (16%), climate change (13%), pesticide application (9%), salinisation (7%), ingress of invasive species (7%), alkalisation (4%), water level fluctuation (2%), boating (2%) and sediment dredging (2%). About 83% of the published case studies reported on the effects of eutrophication alone, whereas 15% documented responses to multiple pressures.

*Table 2. Drivers and pressures checklist for eutrophication in freshwater lakes used in the analysis of pressure-impact relationships. Drivers are underlined and pressures are italicised.*

| **Eutrophication drivers and pressures** | | |
|---|---|---|
| **Primary drivers and pressures** | | **Secondary pressures** |
| Agricultural intensification | Disposal of garden wastes | Sediment dredging |
| Fertiliser run-off | Fertiliser run-off from gardening | Boat disturbance of sediments |
| Animal waste run-off | Inputs from feeding of waterfowl | Metal pollution from mining |
| Soil erosion and losses | Population growth | Invasive species spread |
| Industrial intensification | Sewage discharges | Pesticide discharges |
| Textiles discharges | Waste disposaL | Climate change |
| Food manufacturing dicharges | Construction discharges | Fishery stocking |
| Paper mill discharges | Transport/road run-off | Fish removal |
| Mining discharges | Detergent and soap discharges | Acidification |
| Distillery discharges | Other pressures | Macrophyte harvesting |
| Aquaculture discharges | Waterfowl feaces inputs | Water level management |
| Tourism and recreation | Atmospheric deposition | Waterfowl introduction |
| Food waste disposal | Internal nutrient loading | Extreme weather events |
| Fish stocking | Cyanobacterial N2 – fixation | Industrial thermal-regulation inputs |
| Disturbance of sediments by boats | | |

A range of approaches to eutrophication management were documented. That most commonly reported was a reduction of external nutrient loading (88% of returned publications). In contrast, in-lake management, with or without external loading reduction, received relatively little attention (19% and 6% of returned publications, respectively). Of these, fish biomanipulation (41% of reported in-lake management cases) was the most commonly reported measure, followed by sediment phosphorus (P) capping (16%), drawdown (12%), sediment dredging (13%), flushing (6%), aeration/circulation (5%) and waterfowl/macrophyte biomanipulation (2%).

*Responses*

Responses to eutrophication management, in terms of changes in ecological structure, were assessed for phytoplankton (44% of case studies reporting ecological recovery), macrophytes (15%), zooplankton (14%), macroinvertebrates (13%), fish (12%), waterfowl (2%) and bacterioplankton (<1%). None of the studies reported the restoration of a lake in terms of progress towards Water Framework Directive (WFD) Biological Quality Element (BQE) targets.

The response of in-lake P concentration (the main state change indicator) following catchment nutrient loading reductions was seasonal. Summer concentrations were usually maintained at levels close to pre-management concentrations by P cycling between the sediments and the water column (especially in shallow lakes). In contrast, winter, spring and autumn concentrations tended to fall. There was strong evidence that ecological recovery was being delayed by sediment P processes, especially during summer months. Interactions between seasonally distinct P recovery trajectories and organism growth/colonisation traits (e.g. macrophyte production/colonisation strongest in spring compared to summer during transient period; phytoplankton response weakest in summer compared to winter/spring) were apparent and should be taken into consideration when assessing ecological recovery against WFD targets. Whole-lake manipulation studies have been used to assess the effectiveness of in-lake management techniques (e.g. biomanipulation, sediment dredging, sediment P capping etc.) in reducing the time span of the transient period by controlling sediment P processes. These have met with mixed levels of success.

*Organism groups comparison*

Our meta-analysis identified “impacted” and “recovered” biological communities following eutrophication management. The responses of WFD BQEs (i.e. phytoplankton, fish, benthic macroinvertebrates, macrophytes) were combined with other important groups of organism (e.g. bacterioplankton, zooplankton, waterfowl) to assess the effects of reductions in external pressures on the ecological recovery of lake ecosystems. It was found that alterations in the biological structure of these systems could, potentially, affect ecosystem function and the provision of ecosystem services.

Responses of the bacterioplankton community to eutrophication management are unclear as they have received little attention in the literature. However, the few studies that are available indicated an increase in bacterial abundance and in their relative contribution to energy transfer. In one particular case study these changes were found to be associated, mainly, with a reduction in grazing pressure from Daphnia. It should be noted, however, that this apparently lake-specific response may not be useful for characterising lake response more generally. No changes in production were reported, although it is hypothesised that changes in the quality and quantity of dissolved organic carbon (DOC) associated with eutrophication management may affect community composition and function.

The responses in the phytoplankton community were characterised by strong seasonal changes in community composition and biomass. In general, relative biomass of cyanobacteria decreased, whereas that of diatoms, cryptophytes and chrysophytes increased. The decrease in phytoplankton biomass was typically strongest in spring, winter and autumn in comparison with summer, probably as a result of sediment P processes maintaining TP concentrations at or near to pre-management levels in summer. The responses of heterocystous and non-heterocystous cyanobacteria varied, with non-heterocystous cyanobacteria decreasing in summer and autumn, and heterocystous cyanobacteria increasing in summer and decreasing in spring. A general reduction in diatoms was reported throughout the year, although the reduction was often strongest in spring. This spring reduction was associated with both silica (Si) and P limitation. Although an increase in the chlorophylla:TP concentration ratio can occur as a result of responses to changes in the structure and function of higher trophic levels within a system, this ratio was also found to increase following eutrophication management. Phytoplankton responses were generally associated with reductions in the availability and seasonality of nutrients, resulting in shifts in the competitive advantages of specific phytoplankton taxa. For example, a reported increase in dinophytes in deeper lakes may have been the result of these organisms being capable of migrating vertically through the water column to access water with high TP concentrations in the hypolimnion. Factors confounding the phytoplankton responses included sediment P, N and Si processes, climate change effects (e.g. temperature, precipitation, wind), and fishery management leading to trophic cascades.

The responses of the macrophyte community following eutrophication management were difficult to determine, due to a lack of lake recovery data. However, the most commonly recorded responses have been summarised using long-term data from oligotrophication case studies. These included an increase in colonisation depth, species richness (including relative characean abundance), number of nutrient intolerant species and species distribution. Full recovery of species composition was rarely recorded, potentially as a result of physical barriers to distribution and/or the loss of nutrient intolerant seed banks in cases where eutrophic conditions had been prevalent for many years. Increased water clarity was the most commonly reported driver of macrophyte community responses, although reductions in TN concentrations were also hypothesised to be an important driver in some cases. At a structural level, macrophyte colonisation responses were observed relatively quickly (less than 5 years) after reductions in TP load. However, at a community composition level, recovery timescales for macrophytes were reported to be greater than the transient period estimates outlined above. Factors confounding the responses of macrophytes included grazing by benthivorous fish and birds, distribution barriers and habitat disturbance due to extreme weather events.

The responses of the zooplankton community to reductions in TP inputs were characterised by increases in the relative abundance of Daphnia and other cladocera. This was coupled with an increase in the biomass ratio of zooplankton to phytoplankton. A decrease in zooplankton biomass and an increase in zooplankton species diversity (especially cladocerans) were also reported. These changes were associated with reduced TP concentrations, more shelter from predation associated with increased macrophyte cover, and an improvement in the food quality of the phytoplankton community. The main pressures confounding zooplankton responses included sediment P processes and fishery management. The responses of macroinvertebrate communities to eutrophication management were characterised by an overall reduction in abundance, an increase in species richness and diversity, colonisation of deeper water benthos, and an increase in the chironomid:oligochaeate ratio. These positive responses were associated with a decrease in TP concentrations, a reduction in the delivery of organic detritus to the sediment, an improvement in dissolved oxygen concentrations in the sediments and release from predation pressure. The main pressures confounding recovery of the macroinvertebrate communities were sediment P processes and biomanipulation leading to high organic load to the sediments. Spatially distinct recovery trajectories were reported where shallow, well aerated, zones responded more rapidly than deeper, less aerated, zones.

The responses of fish communities to eutrophication management were generally characterised by a decrease in biomass and a relative increase in piscivorous and percid fish species (especially in shallow lakes). A unimodal response curve of fish species richness to TP concentrations was reported for Danish lakes, with peak richness occurring at 10 400 μg L 1of TP. Long-term studies of eutrophication-oligotrophication have highlighted the following pressure-response pattern in relation to the relative abundance of fish species in response to decreasing TP concentrations: cyprinids-percids-coregonids-salmonids. Responses (typically less than 10 years) in the fish community were commonly observed following external load reduction. These responses were mainly associated with the strengthening of the spring clear-water phase and the provision of better habitat for piscivorous fish. Responses in the fish community did not appear to be strongly confounded by sediment P processes. However, fish stocking and removal practices were reported to confound community responses, especially where these were operated at an industrial scale.

The responses of the waterfowl community to eutrophication management were confounded by interactions between macrophytes and macroinvertebrates, and between herbivorous and benthivorous waterfowl. Few studies were available within which these interactions could be clearly disentangled. An increase in macrophyte cover and community composition (e.g. to favour Chara sp., Elodea spp., Myriophyllum spp. and Potamogeton spp.) was associated with an increase in herbivorous waterfowl (e.g. goldeneye, pochard and coot). However, herbivorous waterfowl were also reported to negatively impact on macrophyte colonisation and community composition. An increase in benthivorous waterfowl was also found to occur as a result of increased macroinvertebrate abundance. The main factor reported to confound waterfowl responses was competition for macrophytes associated with benthivorous fish species.

*Recovery time*

The time span of the transient phase for total phosphorus (TP), phytoplankton, zooplankton and macroinvertebrates response ranged from about 5 years to more than 25 years, whereas that for total nitrogen (TN) was less than 5 years. Rapid recovery (1-3 years) was demonstrated for zooplankton in one case study where multiple management techniques (i.e. diversion of nutrient inputs, biomanipulation and sediment dredging) were applied simultaneously. Fish community responses were typically less than 10 years. Response times for waterfowl following changes in macrophyte extent were reported to range from rapid (i.e. < 1 year) to non-existent.

*Climate / 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 the recovery process. 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. The North Atlantic Oscillation (NAO) was identified as an important driver of weather conditions that are important for maintaining ecological structure in lakes (i.e. precipitation, wind, temperature). 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. Enhanced wind-induced mixing, which leads to habitat disturbance, may also be an important factor.

*Synthesis*

The results from over 40 years (1968-present) of ecological monitoring at Loch Leven (Scotland, UK) were used to identify feedbacks between the components of the DPSIR chain within this lake on the basis of those identified more generally within the literature (Figure 2). These feedback mechanisms demonstrate the complexity of lake eutrophication management and highlight the potential knock-on effects of controlling single pressures within a multi-pressure system.

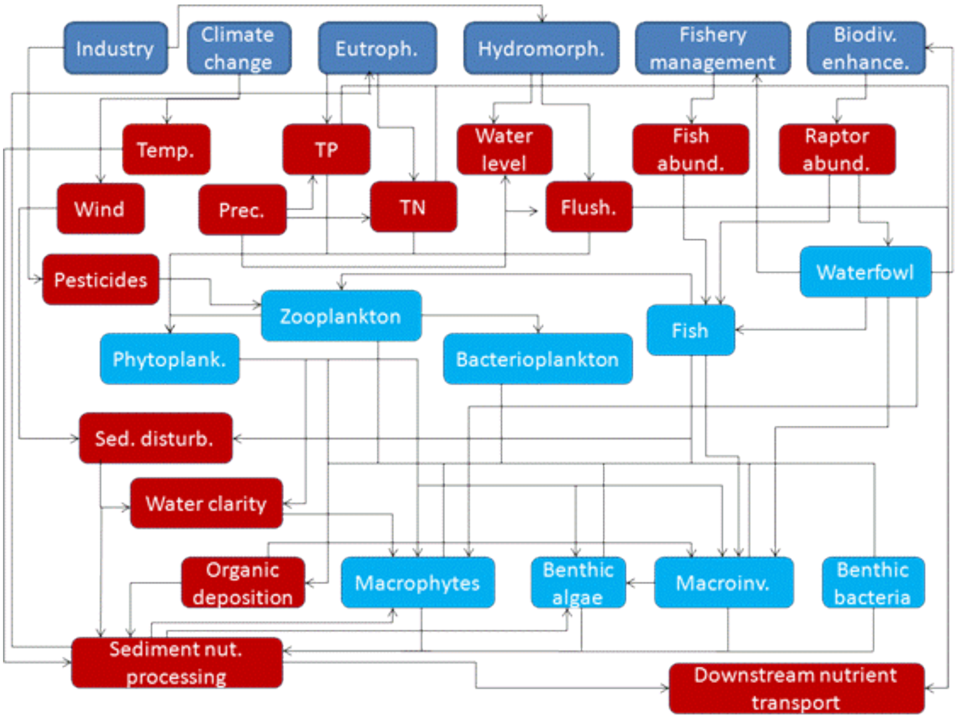



*Figure 2. Inter-relationships reported/hypothesised in the literature between primary (eutrophication) and secondary pressures [dark blue], changes to environmental state [red] and impacts on biological quality elements (BQE) [light blue] in Loch Leven between 1968 and 2010.*

### *3.2 Lakes – acidification*

Introduction

Acidification of surface waters was a severe environmental problem in northern Europe and eastern North America during the second half of the last century, causing a loss of biodiversity and profound alteration of community structure and ecosystem processes (Schindler 1988). With the surge of industrial activity in many developing countries, similar acidification-related problems may arise also in Asian nations in the future (Rhode et al. 1992). Recognition that emissions from burning fossil fuels were resulting in biodiversity loss in surface waters lead to international action plans to protect and restore natural resources (Stoddard et al. 1999). Reductions in the emissions of sulphur dioxide and nitrogen oxide compounds resulted in improved air quality. However, despite improvements in the abiotic environment of surface waters, due decreased deposition of these acidifying compounds (e.g. Stoddard et al. 1999, Skjelkvåle et al. 2005), empirical evidence of biological recovery is equivocal (Skjelkvåle et al. 2003, Stendera & Johnson 2008, Ormerod & Durance 2009, Johnson & Angeler 2010), and several system intrinsic and extrinsic factors have been shown to constrain recovery (climatic variability, drought events, habitat quality, connectivity between habitats, dispersal abilities and species interactions) (e.g. Arnott et al. 2001, Yan et al. 2003, Ormerod & Durance 2009). Since biological recovery is often the ultimate goal of legislative action, not achieving biological objectives means that acidification is still considered as a foremost problem affecting the biodiversity of inland surface waters in northern Europe (Johnson et al. 2003) and elsewhere (e.g. Monteith et al. 2005, Kowalik et al. 2007, Burns et al. 2008).

To achieve biological goals many countries continue to implement large-scale mitigation programmes based on lime application to surface waters and catchments (Henriksson & Brodin 1995, Sandøy & Romunstad 1995). For example, in Sweden, some 7000 lakes and 11000 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). Liming has increased pH and alkalinity in many acidified waters, resulting in improved water quality for aquatic biota. However, studies from Europe and North America have reported mixed results considering the biological responses to liming (Clair & Hindar 2005). In lakes, liming has often, but not always, induced improvements in fish (Appelberg & Degermann 1991, Gunn et al. 1990), phytoplankton (Renberg & Hultberg 1992, Järvinen et al. 1995), zooplankton (Stenson & Svensson 1995, Svensson & Stenson 2002) and benthic macroinvertebrates (Carbone et al. 1998, Persson & Appelberg 2001). Inconsistencies of results among studies may not be surprising, however, abiotic and biotic constraints affect biological recovery in context-dependent ways (Yan et al. 1996, 2003, Binks et al. 2005). These include fluctuations in water chemistry caused by repeated liming and re-acidification events, dispersal capacities of organisms, the characteristics of their habitats, and taxon-specific time lags. Despite these inconsistencies, some generalities arise from liming. Almost all studies have shown that liming-induced community changes are not stable. Strong temporal variability, mediated by a return to an acidified state of the communities when liming was discontinued, characterise biological responses to liming (Clair & Hindar 2005). Therefore, the potential of liming seems limited to partial remediation of acidification impacts, rather than serving as an integral ecological restoration tool that favours the long-term recovery of desired ecosystem structural and functional aspects. Thus, the ecological benefits and economic costs arising from liming deserve critical evaluation, in particular with the environmental benefits that would be obtained from natural recovery without management intervention.

Acidification and recovery of surface waters are landscape-level processes that operate at broad spatial scales. Thus, community dynamics in acidified and circumneutral lakes may be similar as a result of these broad-scale processes. However, despite the latter lake type being better buffered from direct acidification impact, due to a higher acid neutralizing capacity, they may show ecological responses similar to those in acidified lakes, which may be due to the alteration of integral biogeochemical processes resulting from regionally decreased acid deposition. Recent studies provide empirical evidence of dynamical responses of circumneutral and acidified lakes. Evans et al. (2006) suggested that increased concentrations of dissolved organic carbon in surface waters are a result of reduced sulphate in soil solution because of decreasing S deposition. Erlandsson et al. (2008) have shown that this decreasing sulphate deposition synchronizes the oscillation of organic matter content across Swedish streams on a decadal scale. Stendera & Johnson (2008) observed dynamic responses of littoral assemblages to decreased acidification in acidified and circumneutral lakes, and suggested that the signs of “recovery” are detectable in both lake types. These results suggest that the responses of the abiotic and biotic lake environment to reduced acid deposition can be synchronized, thereby further complicating the detection of a recovery signal in acidified lakes.

Recent research has highlighted the link between landscape level changes in water quality resulting from recovery from anthropogenic acidification (increased brownification of surface waters; Erlandsson et al. 2008) and biological invasions. Thus, recovery from acidification may have negative side effects and pose important management challenges. For example, the raphidophycean flagellate Gonyostomum semen (Ehrenberg) Diesing shows an increased incidence of bloom formation across boreal lakes in recent decades (Eloranta and Jarvinen 1991, Lepistö et al. 1994), and this range expansion seems to be favoured by an increasing brownification of lakes (Eloranta and Räike 1995, Findlay et al. 2005, Rengefors et al. 2008). G. semen qualifies as an invasive species according to the definition by Valéry et al. (2008). During bloom formation it can dominate the phytoplankton community by as much as 98% for extended periods (Le Cohu et al. 1989). Furthermore, the incidence and duration of algal blooms is likely to increase with global warming in marine (Rabalais et al. 2009) and freshwater ecosystems (Vilhena et al. 2010), and algal blooms can be considered ecosystem-level perturbations that affect ecosystems structure, function and services (Pickhardt et al. 2002, Rondel et al. 2008). While only a few studies have assessed the impacts of G. semen blooms on structural and functional food web properties in lakes (e.g., Angeler et al. 2010, C. Trigal, D.G. Angeler, T. Vrede, unpublished manuscript), a reduction of recreational services (causing allergic skin reactions in swimmers) and clogging of filters in water treatment plants have been documented (Cronberg et al. 1988; Hongve et al. 1988). The Swedish Environmental Protection Agency therefore treats this alga as a noxious species, and acknowledges the need for more management information. To counteract the negative ecological effects of algal blooms, nutrient reduction schemes have often proved useful as a management tool (Jeppesenet al. 2007ab), but it remains to be assessed whether this strategy will work to control Gonyostomum.

To increase our understanding about integral and transcending ecological responses to decreased acidification a quantification of these effects in the environment is required. Here we summarize the current state of research regarding impact-recovery dynamics in boreal lakes. This review focuses on research published in the scientific and grey literature and ties it to recent unpublished research by the authors of this report. This unpublished research is based on exhaustive analysis of long-term data bases with a good spatiotemporal resolution, which belong to the Swedish Environmental Protection Agency. Based on the review of these results constraints to recovery could be identified that helped us formulate a novel hypothesis framework. Through the formulation of complementary hypotheses a more mechanistic testing of factors that can constrain the recovery process is facilitated.

We specifically test the hypothesis that:

1) Groups of species within a community are differently affected by acidification. This hypothesis is based on the notion that individual species have different sensitivities to acidity, and patterns could emerge at the community level that reflects these different sensitivities. That is, species shall aggregate in different species groups within a community, and these species groups are predicted by different degrees by acidification-related variables. Determining the "species-group-specific" imprint of acidification has management relevance because it allows to numerically quantifying the importance of this anthropogenic stressor in the communities. It also allows a more accurate determination of harmonic responses of structural and functional (i.e. feeding groups) variables to decreased acidification.

2) Temporal patterns in the abiotic and biotic lake environment are spatially congruent. Given the synchronisation in the abiotic environment resulting from reduced acid deposition observed in a previous study on streams (Erlandsson et al. 2008), we expect temporally coherent dynamics in the physicochemical environment of lakes as well. These coherent dynamics in the abiotic environment contribute to synchronisation of community dynamics in lakes.

3) Recovery from acidification facilitates the spread of invasive species. Given the regionally increased water colour in lakes, which has been associated with decreasing acid deposition, we expect that the increased incidence and bloom formation of the nuisance, invasive flagellate Gonyostomum semen, is associated with these changes in the abiotic environment. Such negative side effects of recovery may challenge management, because the resilience of undesired lakes states (bloom formation) may be increased because of such landscape level changes.

4) Liming does not achieve the desired ecological goals. This hypothesis emerges as a consequence of the previous conjectures. Coherent temporal dynamics of lakes with contrasting ecological disturbance regimes suggest that an evaluation of the success of liming can be constrained by a traditional reference-site approach. That is, if limed lakes show similar temporal dynamics as circumneutral or acidified reference lakes, the mitigation potential of liming may not be accurately assessed. Furthermore, evidence exists that liming may create spatiotemporal windows that open invasion opportunities for G. semen (Angeler & Goedkoop 2010). Thus, the combined action of species invasions, likely in combination with other anthropogenic stressors, may even further complicate the assessment of liming outcomes.

*Responses: temporal coherence in the abiotic and biotic environment of acidified and circumneutral lakes*

Analysing twenty-year time series (1988-2007) of physical and chemical variables through a repeated measures analysis of variance including 4 acidified lakes and 8 circumneutral lakes, showed similar time trends in acidified and circumneutral lakes. These variables increased or decreased over the study period, and generally showed a stronger interannual variability in acidified lakes relative to circumneutral lakes (Figure 3). Water temperature, pH, alkalinity and ammonium-N showed no significant treatment x time interaction, suggesting temporally coherent dynamics in acidified and circumneutral lakes (Figure 3, Table 3). However, Secchi depth, electrical conductivity, sulphate, total

P, water colour, TOC and the measures of integral water quality (based on a characterisation through non-metric multidimensional scaling ordinations [MDS]) showed differences in their temporal patterns, resulting in a significant interaction terms in the rm-ANOVA (Table 3; Figure 3).

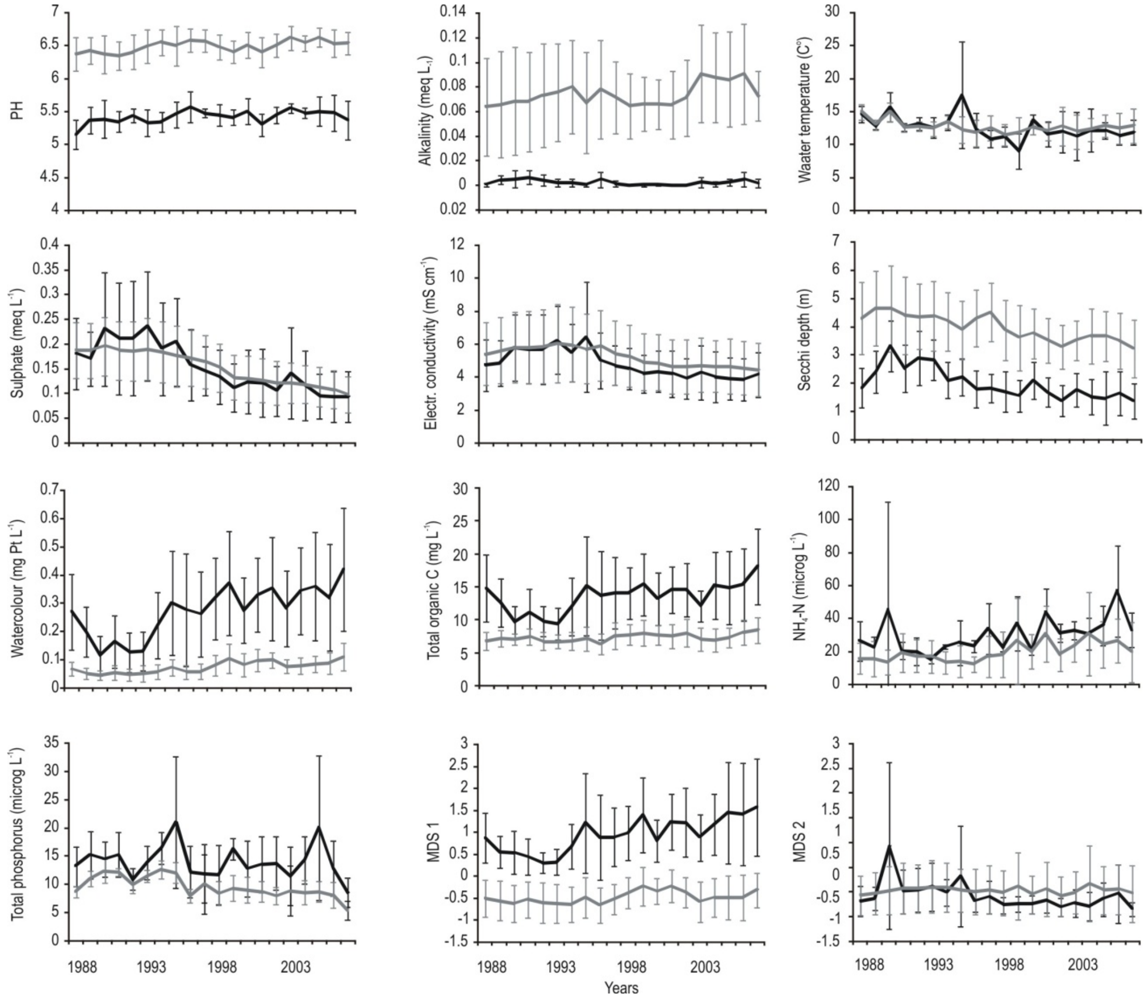



*Figure 3. Temporal trends of physical and chemical variables in acidified (black lines) and circumneutral lakes (grey lines) in southern Sweden. Shown are the means and standard deviations of 4 (acidified) and 8 (circumneutral) lakes, respectively.*

Total macroinvertebrate abundance and species richness increased over the study period in both lake types (Figure 4). Likewise, Shannon entropy showed a slight increase in both lake types; however, trends were weaker compared to species richness and total abundance (Figure 4). Community evenness also showed different degrees of fluctuations in both lake groups (Figure 4). Multivariate measures of community structure (MDS dimensions 1 and 2) showed a similar pattern of temporal change between lake types, independent of whether structure was analysed with compositional or abundance data. The multivariate temporal patterns were similar and overlapping especially along MDS 1, while differences in community structure were better captured along MDS 2 (Figure 4). No significant treatment x time interactions were detected in the ANOVA models of all metrics (Table 4), highlighting that the macroinvertebrate community structural change observed was temporally coherent between both lake types.

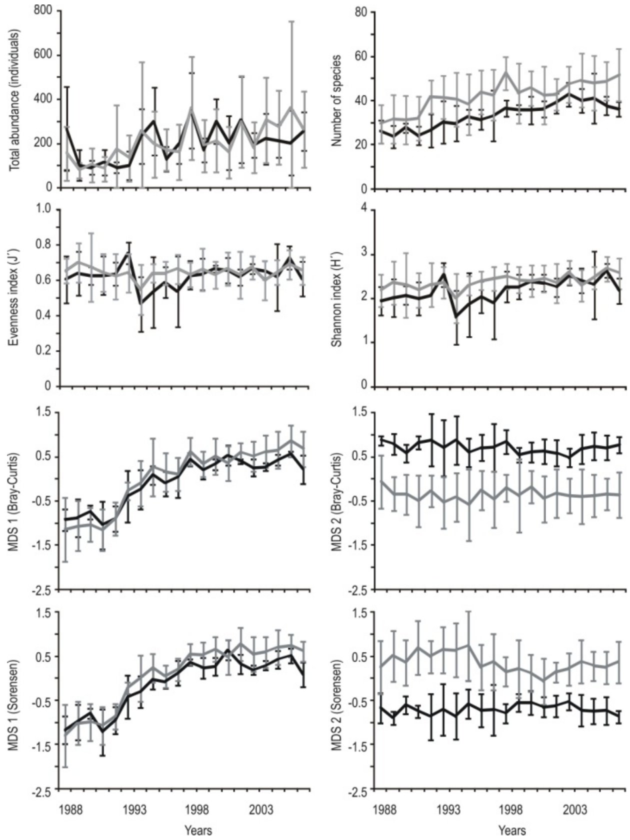



*Figure 4. Temporal trends of univariate community metrics (total abundance, number of species, community evenness, Shannon entropy) and multivariate community similarity (MDS) based on incidence (Sorensen similarity) and abundance (Bray-Curtis similarity) of littoral macroinvertebrates in acidified (black lines) and circumneutral lakes (grey lines) in southern Sweden. Shown are the means and standard deviations of 4 (acidified) and 8 (circumneutral) lakes, respectively.*

*Table 3. Results from repeated measures ANOVA contrasting physical and chemical variables between acidified and circumneutral lakes. Shown are degrees of freedom (df), mean squares (MS), F-ratios, and P levels. Abbreviations: MDS, water quality similarity determined with nonmetric multidimensional scaling ordinations (see methods for details). Shown are similarities among MDS dimensions 1 and 2 of the two-dimensional ordination solution.*

| Dependent vars. | Treatment (df 1, 10) | | | Time (df 19, 190) | | | Treatment x Time (df 19, 190) | | |
|---|---|---|---|---|---|---|---|---|---|
| | MS | F | P | MS | F | P | MS | F | P |
| Secchi depth | 2.700 | 18.70 | 0.002 | 0.033 | 13.30 | <0.001 | 0.007 | 2.99 | <0.001 |
| Water temp. | 0.002 | 0.09 | 0.775 | 0.016 | 5.03 | <0.001 | 0.005 | 1.62 | 0.052 |
| pH | 0.241 | 135.7 | <0.001 | <0.001 | 4.57 | <0.001 | <0.001 | 0.99 | 0.476 |
| Electr. Cond. | 0.036 | 0.09 | 0.771 | 0.225 | 31.02 | <0.001 | 0.002 | 2.66 | <0.001 |
| Alkalinity | 0.05 | 19.98 | 0.001 | <0.001 | 1.61 | 0.055 | <0.001 | 1.24 | 0.227 |
| Sulphate | <0.001 | <0.001 | 0.997 | 0.002 | 34.82 | <0.001 | <0.001 | 2.24 | 0.003 |
| Ammonium-N | 2.08 | 4.07 | 0.071 | 0.118 | 5.06 | <0.001 | 0.024 | 1.04 | 0.422 |
| Total P | 1.053 | 8.06 | 0.017 | 0.058 | 10.55 | <0.001 | 0.011 | 1.95 | 0.013 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Water colour | 0.267 | 19.89 | 0.001 | 0.004 | 21.26 | <0.001 | 0.001 | 7.82 | <0.001 |
| Total org. C | 2.773 | 15.82 | 0.002 | 0.020 | 9.17 | <0.001 | 0.009 | 3.96 | <0.001 |
| MDS 1 | 108.03 | 25.76 | <0.001 | 0.624 | 7.61 | <0.001 | 0.261 | 3.18 | <0.001 |
| MDS 2 | 0.583 | 0.17 | 0.689 | 0.252 | 2.56 | <0.001 | 0.212 | 2.15 | 0.005 |

*Table 4. Results from repeated measures ANOVA contrasting macroinvertebrate community metrics between acidified and circumneutral lakes. Shown are degrees of freedom (df), mean squares (MS), F-ratios, and P levels. Abbreviations: MDS, community similarity determined with nonmetric multidimensional scaling ordinations based on quantitative abundance data (BC, Bray-Curtis similarity) and incidence data (Sør, Sørensen similarity). Shown are community similarities among MDS dimensions 1 and 2 of the two-dimensional ordination solution.*

| Dependent vars. | Treatment (df 1, 10) | | | Time (df 19, 190) | | | Treatment x Time (df 19, 190) | | |
|---|---|---|---|---|---|---|---|---|---|
| | MS | F | P | MS | F | P | MS | F | P |
| Total abundance | 0.176 | 0.18 | 0.677 | 0.042 | 4.71 | <0.0001 | 0.042 | 0.63 | 0.882 |
| Taxon richness | 0.461 | 5.98 | 0.035 | 0.058 | 6.44 | <0.0001 | 0.006 | 0.65 | 0.866 |
| Evenness index | 0.002 | 0.39 | 0.542 | 0.001 | 2.31 | 0.0023 | <0.001 | 1.04 | 0.419 |
| Shannon index | 0.043 | 2.50 | 0.145 | 0.008 | 3.36 | <0.0001 | 0.002 | 0.90 | 0.587 |
| MDS 1 (BC) | 0.588 | 0.79 | 0.395 | 4.108 | 33.05 | <0.0001 | 0.109 | 0.88 | 0.604 |
| MDS 2 (BC) | 60.37 | 26.9 | <0.001 | 0.099 | 0.76 | 0.748 | 0.055 | 0.42 | 0.985 |
| MDS 1 (Sør) | 1.820 | 5.84 | 0.036 | 4.459 | 41.20 | <0.0001 | 0.105 | 0.97 | 0.496 |
| MDS 2 (Sør) | 59.47 | 32.9 | <0.001 | 0.109 | 0.899 | 0.695 | 0.201 | 1.48 | 0.951 |

Pearson correlation relating temporal patterns to multivariate characterisation of macroinvertebrate community structure with environmental variables, revealed the importance of decreasing sulphate concentrations for the observed temporal trends, independent of whether compositional (ordinations based on the Sørensen similarity index; grey bars in Figure 5) or abundance data (ordinations based on the Bray-Curtis index; black bars in Figure 5) were analysed. The imprints of decreasing Secchi depth, electrical conductivity, and increasing water colour were also evident across almost all lakes (Figure 5). Associations of total phosphorus, pH, total organic C, alkalinity and ammonium-N with community structural change were lake-specific and did therefore not contribute to a regionally coherent pattern of change (Figure 5). Also, variables related to climate were less important for explaining community change; temperature showed no consistent correlations across lakes and the NAO winter index was not significantly correlated with the multivariate time trends.

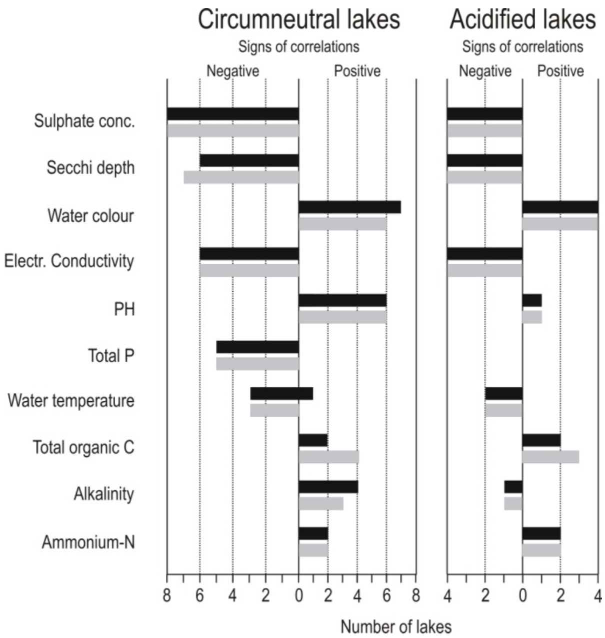



*Figure 5. Overview of results from correlation analyses relating physicochemical variables to MDS dimension 1 of the macroinvertebrate ordinations. Grey and black bars show the number of lakes that showed significant correlations between physicochemical variables and MDS ordinations based on Sørensen-similarity and Bray-Curtis-similarity, respectively.*

*Organism groups comparison: Groups of species within a community are differently affected by acidification*

To address the hypotheses that species groups within lake communities respond distinctly to the imprints of acidification we used a multivariate time series modelling approach (i.e. redundancy analysis where time is modelled with a principal coordinate of neighborhood matrices approach (RDA-PCNM; Angeler et al. 2009). This method identifies species with similar temporal trends in time series and lumps them into species groups based on these trends. The modelled species groups that are obtained by this technique can the be related to abiotic variables, for example through Generalized Linear Models (GLM), allowing us to quantify the magnitude by which these modelled species groups are responding to acidification-related, or other abiotic variables.

We used twenty-year times series (1988-2007) of macroinvertebrate and phytoplankton communities that were sampled in twenty-six lakes that are distributed across Sweden. Regarding the macroinvertebrates, theRDA-PCNM approach revealed significant temporal structure in all of the twenty-six lakes (Table 3). The models identified two species groups with contrasting temporal structure. The patterns associated with the first group of species across all lakes were the most important, explaining on average over 50% of the variance in the models. The patterns explained by the second species group explained on average less than 30% of the adjusted variance in all macroinvertebrate communities across lakes (Figure 6).

Regarding phytoplankton, significant temporal structure was revealed in only 19 of the 26 studied lakes. As was the case with the macroinvertebrates, two species groups were revealed in those lakes with significant temporal structure. The temporal patterns associated with the first species group of phytoplankton explained on average over 60 % of the variance in the models, while patterns associated with the second species group explained on average ca 30% of the variance across lakes.

As for the macroinvertebrates, there was a directional temporal change associated with the first species group of phytoplankton across lakes (Figure 6). The temporal patterns associated with the second species groups showed patterns of change without a directional component (Figure 6).

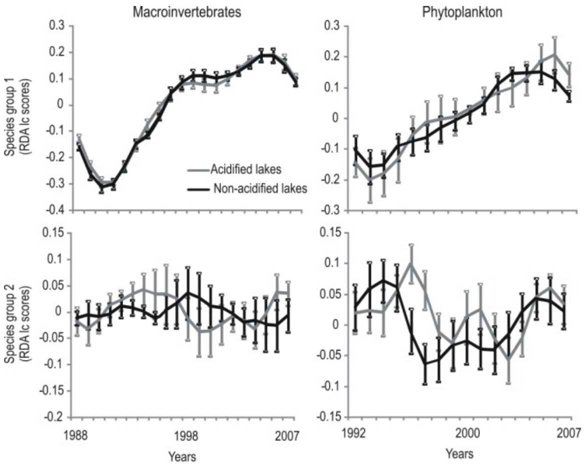



*Figure 6. Temporal trends of species groups associated with RDA axes 1 and 2 for macroinvertebrates (1988 – 2007) and phytoplankton (1992 – 2007). Shown are the overall trends (means ± standard errors) from acidified (grey lines) and non-acidified (black lines) lakes with significant temporal structure.*

Calculating the temporal trends of functional feeding groups for macroinvertebrates (gatherers/collectors, parasites, woodeating taxa, predators, miners, active and passive filter feeders, shredders and grazers/scrapers) and phytoplankton (autotrophs, mixotrophs, heterotrophs), we found that the temporal trends at the functional level rarely tracked the changes observed at the structural level (Figure 7).

Generalized linear models revealed that the temporal trends of macroinvertebrates and phytoplankton species groups were predicted generally by different sets of environmental variables (Table 4). For the macroinvertebrates, almost all lakes showed the clear imprints of SO4 concentrations in the dynamics of species group 1. In addition to SO4, the NAO winter index explained the temporal dynamics of species group 1 in some lakes. In a few lakes total phosphorus concentrations explained the temporal patterns of the second species group; otherwise the species group 2 was not significantly related to environmental variables in most lakes. Regarding the phytoplankton assemblages, the temporal dynamics of species groups were explained by SO4, temperature, NAO, TOC, TP, or different combination of these five variables (Table 5). The imprints of SO4 were captured in species group 1 of phytoplankton in all acidified and most non-acidified lakes. As was the case for macroinvertebrates, the second species group within the phytoplankton communities was unrelated to environmental variables in many lakes. GLMs also revealed that functional feeding groups were not significantly related to the structural changes associated with both species groups in phytoplankton and macroinvertebrates.

*Table 5. Summary of generalized linear model (GLM) results. Shown are environmental variables that explained the temporal dynamics of different species groups (associated with canonical axes 1 and 2, respectively) revealed by time series modelling of macroinvertebrates and phytoplankton. Degrees of freedom (df), F-ratios and significance levels for each variable are given in parentheses (**

*P < 0.05; ** P<0.01, *** P < 0.001). Abbreviations: TP, total phosphorus, TOC, total organic carbon; Temp, temperature; SO4, sulphate concentration; NAO, North Atlantic Oscillation winter index; Ns, GLM not significant; ---, no significant temporal structure revealed by time series modelling.*

| Lakes | Macroinvertebrates<br>Species group 1 | (df 19)<br>Species group 2 | Phytoplankton<br>Species group 1 | (df 15)<br>Species group 2 |
|---|---|---|---|---|
| ***Non-acidified lakes*** | | | | |
| Mäsen | $SO_4$ (30.7***) | Ns | --- | --- |
| Stora Skärsjön | $SO_4$ (53.8***) | Ns | $SO_4$ (18.6***) | Ns |
| Brännträsket | $SO_4$ (30.5***) | Ns | TP (6.1*)<br>TOC (7.1*) | $SO_4$ (17.6**) |
| Fräcksjön | $SO_4$ (63.1***) | Ns | $SO_4$ (5.9*) | $SO_4$ (11.4**)<br>TOC (13.5**) |
| Stensjön | $SO_4$ (19.8***) | Ns | NAO (5.5*) | Temp (8.2*)<br>TOC (6.0*) |
| Tväringen | $SO_4$ (31.6***) | Ns | TOC (7.5*) | $SO_4$ (23.6***)<br>Temp (7.3*) |
| Skärgölen | $SO_4$ (20.7***) | $SO_4$ (6.6*)<br>Temp (5.4*) | $SO_4$ (69.4***)<br>NAO (5.9*) | Ns |
| Remmarsjön | $SO_4$ (9.5**)<br>NAO (12.6**) | Ns | $SO_4$ (23.5***)<br>TOC (7.7*) | --- |
| Stora Envättern | $SO_4$ (22.7***)<br>NAO (18.4***) | $SO_4$ (5.8*) | NAO (15.4**)<br>TOC (16.7**) | $SO_4$ (9.0**) |
| Jutsajaure | $SO_4$ (5.2*)<br>NAO (10.5**) | $SO_4$ (5.7*) | $SO_4$ (17.0**)<br>Temp (7.1*) | TOC (15.3**) |
| Fiolen | $SO_4$ (46.8***)<br>NAO (7.3*) | Ns | Ns | Ns |
| Allgjuttern | $SO_4$ (100.4***)<br>Temp (5.0*) | Ns | $SO_4$ (53.1***) | Ns |
| Abiskojaure | TP (12.6**) | $SO_4$ (53.1***) | --- | --- |
| Humsjön | NAO (6.2*)<br>TOC (4.7*) | $SO_4$ (10.3**) | $SO_4$ (31.0***) | TP (4.9*) |
| Stor Thjulträsket | NAO (4.8*) | Ns | Ns | --- |
| ***Acidified lakes*** | | | | |
| Brunnsjön | $SO_4$ (26.6***) | Ns | $SO_4$ (46.2***) | Ns |
| Bysjön | $SO_4$ (39.1***) | Ns | --- | --- |
| Hagasjön | $SO_4$ (20.0***) | Ns | $SO_4$ (20.3***)<br>TOC (7.4*) | Ns |
| Harasjön | $SO_4$ (62.3***) | Ns | $SO_4$ (11.3**)<br>TOC (6.7*) | Ns |
| Sännen | $SO_4$ (30.6***) | Ns | --- | --- |
| Övre Skärsjön | $SO_4$ (51.8***) | Ns | --- | --- |
| Grissjön | $SO_4$ (43.8***) | TP (15.4**) | $SO_4$ (40.6***) | Temp (7.7*) |
| Storasjö | $SO_4$ (72.9***) | TP (8.5**) | Ns | $SO_4$ (5.0*) |
| Rotehogstjärnen | $SO_4$ (25.8***)<br>NAO (19.4***) | Ns | --- | --- |
| Älgarydssjön | $SO_4$ (29.0***)<br>NAO (19.8***) | TP (9.7**) | $SO_4$ (33.8***) | Ns |
| Härsvatten | $SO_4$ (9.4**)<br>NAO (13.5**) | $SO_4$ (17.8***)<br>Temp (15.2*) | --- | --- |

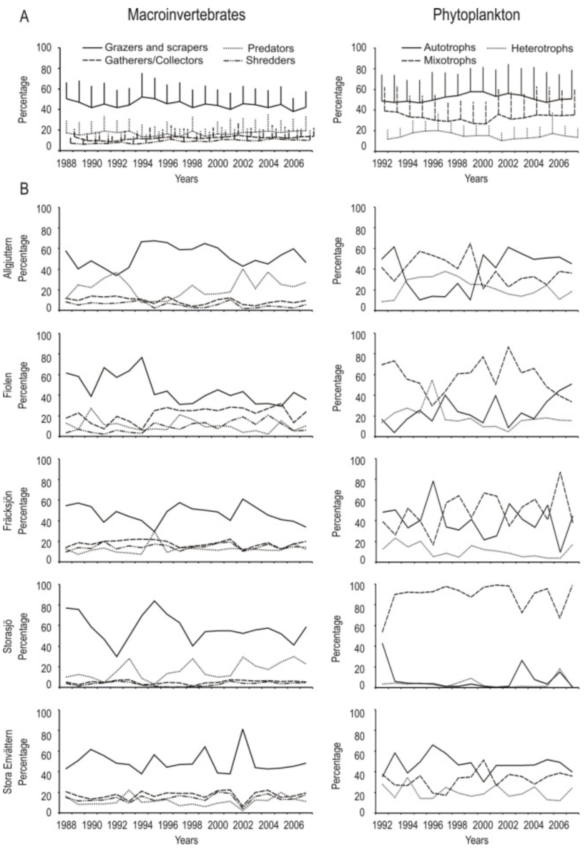


*Figure 7. Temporal trends of functional groups of invertebrate and phytoplankton communities. Shown are the overall trends (means ± standard deviations) from lakes with significant temporal structure (n = 26, invertebrates; n = 19, phytoplankton) (A), and the trends in selected lakes (B).*

<u>Recovery from acidification facilitates species invasions</u>

Furthermore, we quantified biomass aggregation patterns of the invasive, nuisance flagellate, *Gonyostomum semen*, using simulation models and cluster analysis and analysed the incidence of this alga during an eleven-year period between (1997-2007) in 78 lakes distributed throughout Sweden. We found that the number of biomass aggregation groups increased from 2 to 6 groups over the study period; this increase was significant (Figure 8A). Concomitantly, *G. semen* was increasingly detected in lakes resulting in a significant decrease of *G. semen*-free lakes over time (Figure 8B). The decreasing number of *G. semen*-free lakes was correlated with the increase in biomass aggregation groups (Spearman rho -0.7, P = 0.016, n = 11). Spearman rank correlations also showed that the increased number of biomass groups, but not the decreasing number of *G. semen*-free lakes, were associated with decreased sulphate concentrations (Spearman rho -0.75, P = 0.01, n = 11), increases in alkalinity (Spearman rho 0.6, P = 0.05, n = 11), and the temporal variability of water colour (Spearman rho 0.66, P = 0.03, n = 11) across lakes over the study period. The regional temporal patterns of these variables are shown in Figures 9A-C. Thus, the patterns of increased biomass aggregations have an underlying environmental component (i.e. it is related to factors that capture the effects of decreased acid deposition) that depends on the regional occurrence of *G. semen* in the lakes.

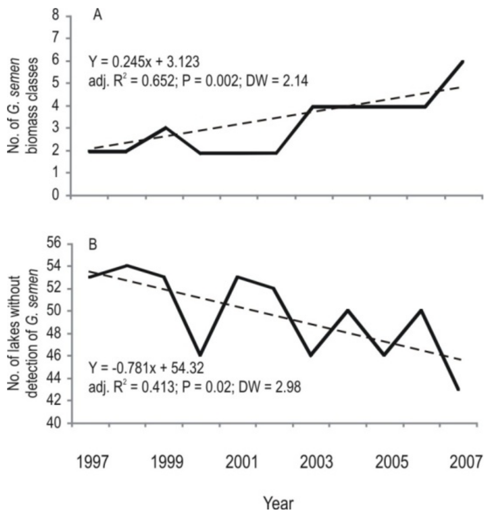



*Figure 8. Regression results of temporal change of number of resolved biomass classes of G. semen (A) and detection of G. semen in lakes over time (B). DW, Durbin-Watson statistic (values >1 indicate low risk of biased results due to temporal autocorrelation of residuals).*

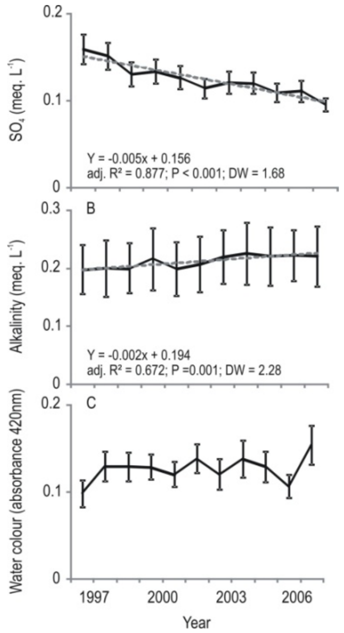



*Figure 9. Temporal trends of sulphate concentrations (A), alkalinity (B), and water colour (C) between 1997 and 2007. Shown are the means ± standard errors of 75 lakes. Significant temporal changes are indicated by the linear trends lines and regression details.*

<u>The efficiency of liming as a mitiation tool</u>

A recent study compared congruencies in biological responses to lake liming using phytoplankton, zooplankton, and fish in pelagic habitats and benthic macroinvertebrates in littoral, sublittoral and profundal habitats (Angeler & Goedkoop 2010). The study was based on 4 acidified, 7 circumneutral and 12 limed lakes. Non-metric multidimensional scaling analyses (NMDS) were used to determine community structure in these lake types over a 5 year periods (2000-2004).

The NMDS analyses showed a clear separation of communities of acidified, circumneutral, and limed lakes (Figure 8), and a complementary analysis of similarity (ANOSIM) indicated a among-lake difference in community similarity ($P < 0.05$). NMDS analyses of functional feeding groups show a good overall agreement with the structural analyses, except that zooplankton and littoral macroinvertebrates in limed and circumneutral lakes were functionally similar.

The NMDS ordinations based on community structure showed two types of patterns. First, zooplankton and macroinvertebrate communities of limed lakes occupied intermediate positions between acidified and circumneutral lakes (Figure 10), resulting from an incomplete species gain during the recovery process. The second pattern revealed was that phytoplankton and fish communities in limed lakes were positioned off the gradient between acidified and circumneutral conditions. This pattern was due to a relatively high proportion of species unique to each lake type and the relative contribution of species shared by lake types. For example, the raphidophycean flagellate *Gonyostomum semen* had a high contribution in limed (ca. 42 %) and acidified lakes (ca. 54%), while this species occurred only marginally in circumneutral lakes. Similarly, the fish communities of limed lakes were characterized by the presence of *Salmo trutta* and *Salvelinus alpinus* and the absence of *Tinca tinca* relative to circumneutral lakes, leading to ca. 28% dissimilarity between lake types.

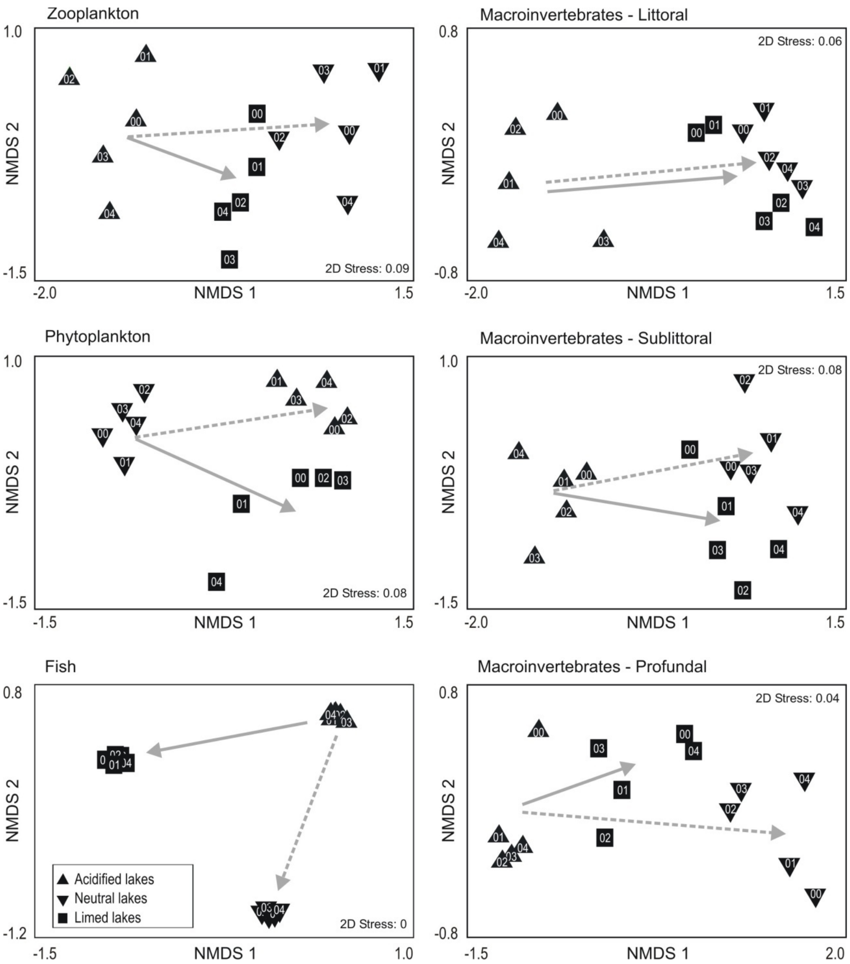



*Figure 10. NMDS ordinations showing temporal trends of phytoplankton, zooplankton, fish and macroinvertebrate (in three habitats) communities in acidified, circumneutral and limed lakes between 2000 [00] and 2004 [04]. Grey full arrows indicate desired recovery trajectories from acid towards circumneutral conditions. Grey dotted arrows indicate observed community trajectories resulting from liming.*

*Synthesis*

Several important management implications derive from this study. The ultimate goal of legislative action to counteract the negative ecological effects of cultural acidification is to achieve biological recovery objectives in acidified lakes. Our results show that the effects of reduced sulphate deposition go beyond recovery. While reduced emission of acidifying compounds to the atmosphere was clearly beneficial for aquatic ecosystems, our study demonstrates overarching ecological consequences that affect similarly the dynamics of lake water quality, and consequently the temporal patterns of communities in acidified and circumneutral lakes. As a result, discerning recovery signals in acidified lakes from broad-scale background variability, resulting from reduced acid deposition across lakes, is difficult because of the spatially synchronized dynamics of acidified and circumneutral lakes. The observed temporally coherent trends across lakes may contribute to our understanding of why recovery from acidification has thus far been equivocal (Skjelkvåle et al. 2003, Stendera and Johnson 2008, Ormerod and Durance 2009, Johnson and Angeler 2010), and desired goals often not achieved. Identifying temporal coherence as a constraint to recovery adds to several factors which have been shown to limit biological recovery from acidification. These include several system intrinsic and extrinsic factors, including climatic variability, drought periods, poor habitat quality, limited connectivity between habitats, dispersal abilities and biological interactions have been suggested to constrain recovery (Arnott et al. 2001, Yan et al. 2003, Ormerod and Durance 2009). Given that our results demonstrated that changes in the abiotic environment that are due to reduced acid deposition (Evans et al. 2005, 2006, Erlandsson et al. 2008) consistently correlated with the spatial synchrony across lakes, the evaluation of biological recovery from acidification using a traditional least-impacted reference site approach can be complicated.

In addition to this landscape-level constraint for evaluating recovery success, our study demonstrates an environmental dilemma whereby the invasion of pest species can further contribute to confounding recovery success. The environmental dilemma described here is mediated by an increased “brownification” of surface waters related to reduced acid deposition (Erlandsson et al. 2008), and correlated with the spread of the bloom-forming G. semen. A recent study has also shown that global climate change can affect the spread and biomass development of G. semen. Longer durations of the vegetation period in northern latitudes cause blooms to persist until autumn (Bloch 2010). Thus, the dilemma documented here likely results from a combination of climate warming and recovery from acidification and is manifested in mutually reinforcing processes that will likely complicate management endeavours. While current management options for controlling algal blooms are mainly based on nutrient reduction schemes (Jeppesen et al. 2007ab), the lack of relationships of nutrient variables with G. semen biomass formation (Angeler et al. 2010, Bloch 2010) suggests that such traditional management tools will unlikely work for G. semen.

Our review also provides insight into the effectiveness of lake liming to mitigate the effects of cultural acidification. The comparative study of multiple communities in limed, acidified and circumneutral lakes by Angeler & Goedkoop (2010) facilitated an important assessment of ecological responses of boreal lakes to management practices. As has been demonstrated for stream ecosystems (McKie et al. 2006, McClurg et al. 2007, Ormerod & Durance 2009), these results show that liming has a limited restoration potential in lakes.

In NMDS ordinations, these differences were indicated by the positions of limed lakes relative to acidified and circumneutral lakes. Phytoplankton and fish communities in limed lakes were positioned away from acidified and circumneutral lakes, whereas the other communities occupied intermediate positions between acidified and circumneutral conditions. The intermediate position of the macroinvertebrate and zooplankton communities of limed lakes illustrated a limited gain of species and different individual biomass patterns relative to circumneutral lakes, while the phytoplankton ordination highlighted additional factors that can confound liming effects. Both limed and acidified lakes showed mass occurrences of the invasive flagellate Gonyostomum semen (Raphidophyceae), suggesting that liming does not counteract other undesired effects resulting from global change. Our results therefore suggest that other forms of anthropogenic stress, including species invasions, will probably lead to novel environmental situations, which increase the uncertainty and predictability of management schemes (Harris et al. 2006). Global climate change is likely to add further confounding factors (Skjelkvåle et al. 2003, Angeler 2007). Thus, managing boreal lakes to address a single environmental problem is likely to become obsolete. Managers are challenged to re-evaluate the benefits of liming and its contribution to broader management schemes that tackle several forms of anthropogenic stress simultaneously.

We finally provide some preliminary reflection concerning ecosystem resilience in face of the results of this review. Functional group attributes were largely uncorrelated with the structural changes over time, suggesting that both invertebrate and phytoplankton communities in boreal lakes can be resilient to environmental change. Considering that the emergent structural and functional attributes mediated by communities indicate ecosystem processes (Allen et al. 2005), our results would suggest that boreal lakes, and the ecosystem services they provide to humans and wildlife, could resist some of the threats arising from global change in the future. Nonetheless, it is critical to understand the roots of this apparent resilience to avoid catastrophic consequences when a resilience threshold is passed and systems collapse. Further research is required for quantifying resilience in a transforming boreal lake landscape which seems to be steered by the consequences of reduced acid deposition.

### *3.3 Lakes & Rivers – acidification*

*Drivers and pressures*

The acidity of rivers and lakes affected by atmospheric sulphur and nitrogen deposition is decreasing over large areas of Europe and North America. Time series data at numerous monitoring sites show increasing pH and acid neutralising capacity and reductions in concentrations of labile aluminium and non-marine sulphate. This is primarily a consequence of reductions in emissions of sulphur. Liming has also resulted in improved water chemistry in Northern Europe although there have been concerns about the value of liming as a long-term management strategy. However, despite widespread improvements in water quality there are other factors which may be limiting chemical recovery at some sites including inadequate reductions in deposition, limited recovery of soil base saturation, continued release of sulphur from catchments soils, increases in the release of nitrate from catchment soils and the influence of other stresses such as forest growth and climate change.

*Responses and organism groups comparison*

There is evidence of biological recovery in some rivers and lakes exhibiting chemical recovery including the reappearance of acid sensitive species and greater diversity in some sites. However, the evidence is geographically inconsistent and often relates to a single biological group. The response of various organism groups also differs as studies looking variously at diatoms, phytoplankton, macrophytes, zooplankton, macroinvertebrates and fish have shown. Generally, the biota at many recovering sites remains impoverished. Much of the attention in such studies continues to be on species compositional changes (highlighting where previously 'lost' species have reappeared or species new to the record have been identified) with indicator species often the focus. There are, however, relatively few accounts of functional responses to recovery from acidification. The effects of biological interactions within altered food-webs in formerly acidified lakes and the effects of the biological and physical factors controlling species dispersal and colonization success remain poorly understood. While some studies have looked at the structural and functional response (for example invertebrate species richness and decomposition of leaf litter) to acidification, few have given consideration to the recovery limb.

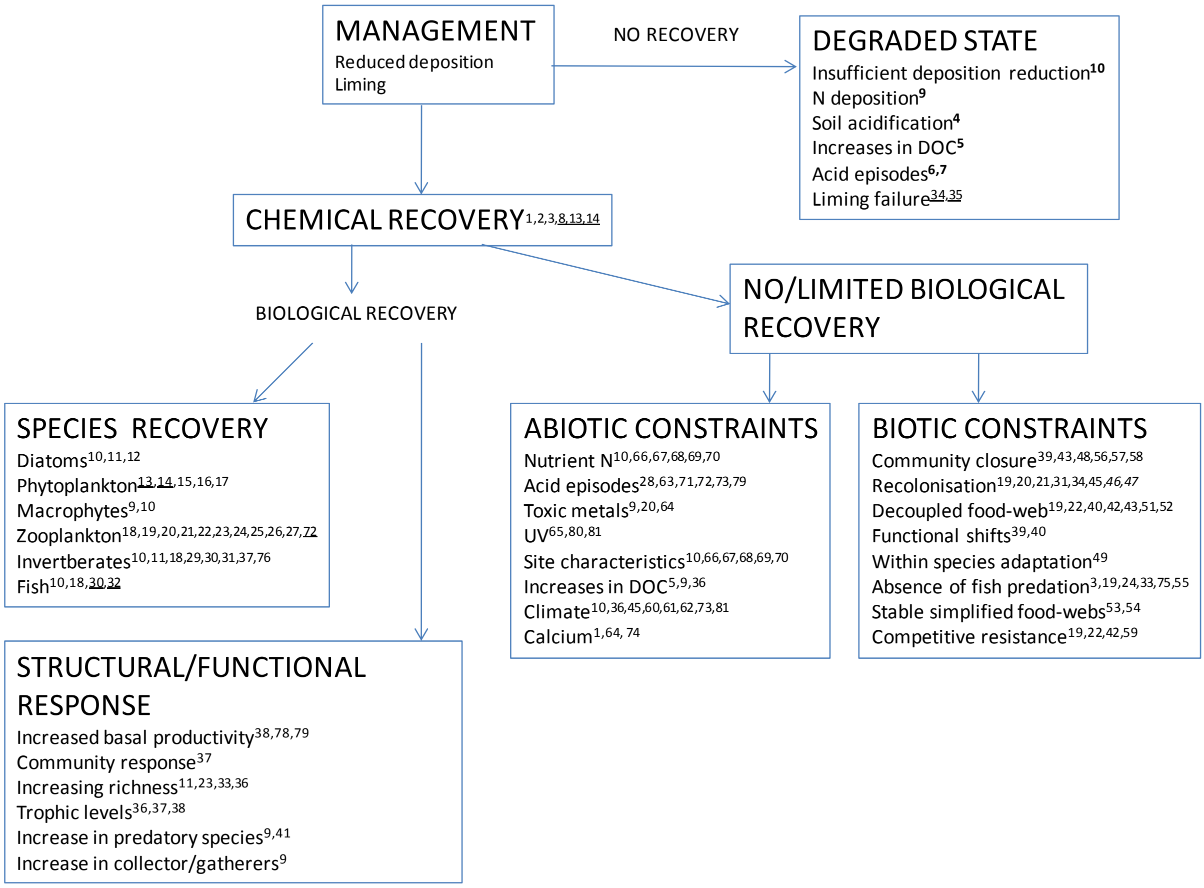
*Synthesis*

The bulk of the evidence to date indicates that biological recovery may be or will be delayed following chemical restoration. The notion that 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 ecological inertia leading to biological resistance to recolonisation (priority effect) 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 resistance to recovery in terms of species composition. Biological resistance 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.

There have been discussions about the value of liming as a long-term management strategy. Chemical improvements following liming have been observed but, despite evidence of some ecological response, it is apparent that liming produces mixed response with regard to biological recovery. Although recovery in some organism groups has been observed following liming-induced reductions in habitat acidity, most studies indicate that the community response to liming may not stable. It has been suggested that this may stem from the episodic nature of the treatment, differences between the 'natural' and liming recovery processes with regard to calcium, the fact that liming doesn't increase regional colonist pools and the rapidity of the chemical response compared with natural recovery. In terms of long-term ecological goals it seems that natural recovery following deposition reductions is preferable to costly liming programmes. What is becoming apparent from the research undertaken to date is that ecological recovery is not likely simply to be a mirror image of the degradation process. Pathways and trajectories of recovery are not always linear (i.e. they display hysteresis). There is evidence in some cases that the ecological recovery trajectory is not tracking back towards a pre-acidification reference but towards a different assemblages compared with the composition change that occurred during the acidification stage. Therefore, the predisturbance conditions may not necessarily be an appropriate recovery target for aquatic ecosystem management. It may be the case that the acidification of freshwaters represents a regime shift in ecosystems that may not simply be reversed.

In numerous cases the data sets used to monitor ecological recovery are not long enough 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.

There appears to be relatively little published on functional responses to recovering surface waters. The review represents a wider summary of literature on biological recovery, how this is manifested and the potential barriers to recovery. Though the topic is acidification in lakes streams were also included as many of the papers looked at streams individually and some looked at streams and lakes. In summary, the schematic representation of the recovery review is as follows:


MANAGEMENT
Reduced deposition
Liming
NO RECOVERY
DEGRADED STATE
Insufficient deposition reduction[10]
N deposition[9]
Soil acidification[4]
Increases in DOC[5]
Acid episodes[6,7]
Liming failure[34,35]
CHEMICAL RECOVERY[1,2,3,8,13,14]
BIOLOGICAL RECOVERY
NO/LIMITED BIOLOGICAL RECOVERY
SPECIES RECOVERY
Diatoms[10,11,12]
Phytoplankton[13,14,15,16,17]
Macrophytes[9,10]
Zooplankton[18,19,20,21,22,23,24,25,26,27,72]
Invertberates[10,11,18,29,30,31,37,76]
Fish[10,18,30,32]
ABIOTIC CONSTRAINTS
Nutrient N[10,66,67,68,69,70]
Acid episodes[28,63,71,72,73,79]
Toxic metals[9,20,64]
UV[65,80,81]
Site characteristics[10,66,67,68,69,70]
Increases in DOC[5,9,36]
Climate[10,36,45,60,61,62,73,81]
Calcium[1,64, 74]
BIOTIC CONSTRAINTS
Community closure[39,43,48,56,57,58]
Recolonisation[19,20,21,31,34,45,46,47]
Decoupled food-web[19,22,40,42,43,51,52]
Functional shifts[39,40]
Within species adaptation[49]
Absence of fish predation[3,19,24,33,75,55]
Stable simplified food-webs[53,54]
Competitive resistance[19,22,42,59]
STRUCTURAL/FUNCTIONAL RESPONSE
Increased basal productivity[38,78,79]
Community response[37]
Increasing richness[11,23,33,36]
Trophic levels[36,37,38]
Increase in predatory species[9,41]
Increase in collector/gatherers[9]

## 3.4 Rivers – hydrology: removal of weirs and dams

*Introduction: drivers and pressures*

Weirs and dams were build for water storage, flow regulation, hydropower and other uses. 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 invertebrates (e.g., 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 (e.g., Bednarek et al. 2001; Hart et al. 2002).

*Responses*

Most studies were conducted about North American streams, and only one restoration study (Tszydel et al. 2009) and two reviews originated from European streams (Schmitt et al. 2005; de Leaniz et al. 2008). All restoration studies compared conditions before and after weir removal (BA), while 12 out of 22 restoration case studies included comparisons of control and impacted sites (BACI). Most of the studies have sampled several hundred meters stretches totalling overall into sections of kilometres.
The removal of dams and its possible ecological impact on riverine organisms has been reviewed by Bednarek (2001), who also presented a series of case studies to underpin the review with real data. Accordingly, several important river characteristics are positively affected by the removal of dams and other transverse structure that cause impoundment. An unregulated flow regime allows the return of lotic and dynamic flow conditions to formerly impounded sections. Bunn and Arthington (2002) stressed the role of flow as a major determinant of physical habitat in streams, which in turn is a major determinant of biotic composition. Acreman and Dunbar (2004) referred to the flow regime required in a river to achieve desired ecological objectives, i.e. the 'environmental flow'. Environmental flow includes floods, medium and low flow, as all elements of a flow regime are considered important (Poff et al. 1997). Low flows provide a minimum habitat for species and prevent invasives, medium flows sort river sediments and stimulate fish migration and spawning, and floods maintain channel structure and allow movement between floodplain habitats (Acreman and Dunbar 2004).
Occasional floods reconnect the aquatic and riparian habitat (Shuman 1995; Jähnig et al. 2009) and backwaters are refilled. Fine materials (e.g., sand, silt, mud) erode and uncover coarser substrata (e.g., gravel, pebble and cobbles), which enhances the overall habitat diversity (Kanehl et al. 1997; Born et al. 1996). The sediment transport also affects habitat diversity further downstream. Dissolved oxygen and water quality improve (Hill et al. 1993), the temperature regime changes (less warming of stagnant water). Bednarek (2001), however, also refers to some negative effects, such as contamination further downstream due to the transport of contaminated sediments or the overall abrasive effect of fine sediment transport. But these adverse effects are considered rather short-term, while improvement will occur in the long-term.
Overall, the changing abiotic conditions improve biodiversity and reproduction of fish. The spawning grounds for salmonid species increase (Iversen et al. 1993), while fish passage is more likely because of the restored longitudinal connectivity. Hence, typical riverine (migrating) fish benefit, while lentic and reservoir-specific species decrease. The maintenance of the longitudinal, but also of the lateral connectivity with the floodplain is essential to the viability of populations of many riverine species (Bunn and Arthington 2002).
As Stanley and Doyle (2003) suggest, weir removal may be best considered as ecological disturbance. Removal of small dams generally transform lentic to lotic river systems upstream of the former dam leading to the reservoir sediment release and a pulse of disturbances to downstream reaches: i.e., temporary increases in suspended and bed sediment loads that will cause short-term reductions in

productivity and possibly diversity (Bednarek 2001). In addition, effects of restoration could be very variable depending on the hydrologic nature of the river (Chaplin, 2003). As a result, the effectiveness of a dam removal, i.e. the recovery of a river from the induced disturbance is expected to be very diverse from a case to another.

The literature provides little information on restoration effectiveness. Indeed, the effectiveness is rarely measured and elements of success are very vague. However, in most cases, negative impacts of weir removal are short term effects (e.g. increase in suspended sediments) while beneficial impacts are long term effects (e.g. increase in flow diversity, connectivity) and that the natural free-flowing state of the river is always regained whereas recovery of BQEs following this habitat shift is more uncertain.

Weir removal is different from the two cases presented before (see Annex 1.10), since it is necessary to distinguish upstream and downstream effects to understand the different processes involved. Indeed, effects of weir removal are quasi inverted upstream and downstream the barrier, such as changes in substrate particle size. In general, however, physical and hydromorphological processes have been frequently investigated in the literature, while there is little information on the impacts of changing habitat conditions on different organism groups.

Altogether, four major cause-effect relationships can be frequently found, the first three of which are also supported by this study, based on the number of references (Figure 11):

- i) the increase of the sediment particle size upstream and the decrease downstream of the former weir due to shifting fines from the former upstream (impounded) section further downstream,(e.g., Chaplin 2003; Cheng et al. 2007),
- ii) the increase of the flow diversity upstream (e.g., Hill et al. 1994),
- iii) the decrease of the water temperature upstream (e.g., Kanehl et al. 1997; Hill et al. 1994), and
- iv) the restoration of the hydro-ecological connectivity (e.g., Poff 1997; Gregory et al. 2002).

Effects of the restoration of the longitudinal connectivity on in-stream plant and animal recovery have been highlighted, for instance, by Bushaw-Newton et al. (2002) and Maloney et al. (2008). In particular, the recovery of migratory fish following a re-establishment of the hydrological connectivity is a common key argument for restoration (Iversen et al. 1993; Poff 1997; but see also de Leaniz et al. 2008 for a more recent summary of findings during the past decade).

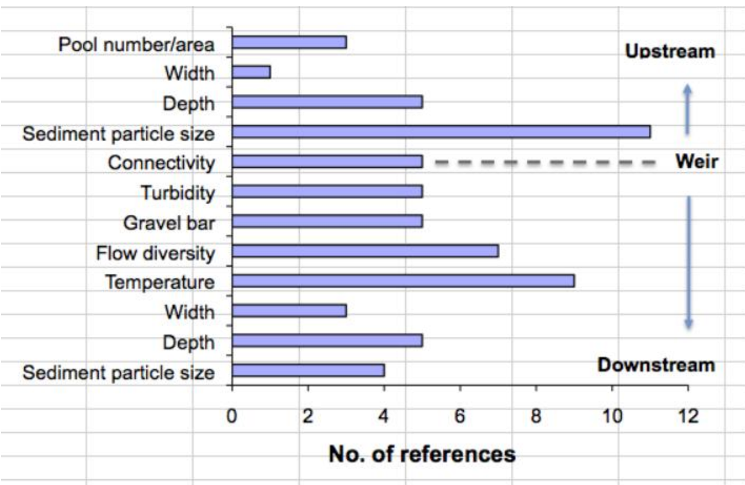


*Figure 11. Environmental State variables most often referred to in the reviewed literature. (Total number of references N = 36).*

Organism groups comparison

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 12). For all organism groups, Impact is usually expressed by community composition and abundance. Measures of species richness and diversity were only used in studies on benthic invertebrates and macrophytes. Some papers also consider effects on community functional metrics such as benthic invertebrate feeding habits (e.g., Maloney et al. 2008) or fish growth (e.g., Schlosser 1982; Harvey et al. 1991). Twelve papers have studied the effects of weir removal on sensitive and tolerant benthic invertebrate taxa, mainly of EPT taxa (Ephemeroptera-Plecoptera-Trichoptera) and the effects of water quality improvement such as the abatement of turbidity and oxygen enrichment (e.g., Orr et al. 2006; Bushaw-Newton et al. 2002). Twenty three studies support the six links related to fish Impact (see Annex 1.14) revealing two main positive effects: the reestablishment of the connectivity and the increase of gravel bar downstream following erosion processes and sediment flush (e.g., Gregory et al. 2002). Eighteen papers support seven links to benthic invertebrates, nine papers of which corroborate the negative impacts of fine sediments on the availability of coarse gravel downstream (e.g., Pollard et al. 2004; Thomson et al. 2005; Orr et al. 2006). The Impact on the macrophyte community is most often associated with changes in channel morphometry (depth, width) and connectivity (e.g., Shafroth et al. 2002), while phytobenthos composition and abundance respond to changes in sediment size and turbidity (e.g., Baattrup-Pedersen et al. 1999; Orr et al. 2006).

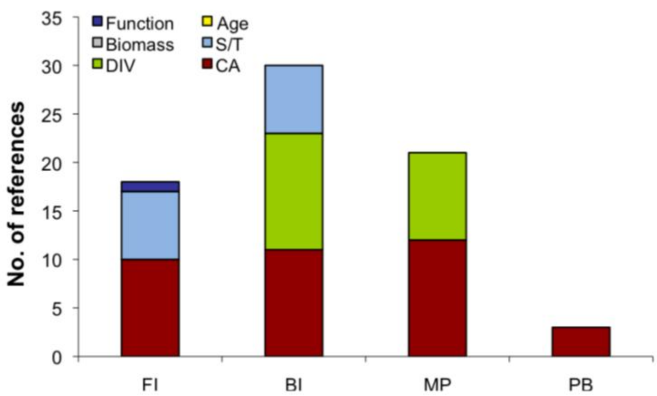



*Figure 12. Number of references addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB).*

Recovery time

The findings presented here are consistent with the conclusions of Doyle et al. (2005): each variable evolves in a specific time scale after weir (dam) removal. The re-establishment of the longitudinal connectivity that allows migratory fish to move is quasi-immediate. Many environmental parameters, such as water temperature and substrate conditions, and the overall water quality may recuperate within a few years. In contrast, biological recovery in general requires several years or even decades 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, the full recovery may take up to 80 years, but the reviewed literature does rarely include monitoring periods longer than five years. The time-scale of recovery after weir removal continues to remain speculative unless long-term monitoring is being conducted.

*Synthesis*

All studies reviewed here provide qualitative analyses, but no reference reported quantifiable results in the sense of statistical (e.g., regression) or mechanistic relationships (Table 6). Nonetheless, sound multivariate analysis (ANOVA, ordination) has been frequently used to detect and identify patterns of biological Impact (e.g., Bushaw-Newton et al. 2002; Pollard et al. 2004; Thomson et al. 2005). Cheng et al. (2007) studied the removal of a dam (height: 2.2 m) on the Sandusky river in Ohio and showed that bed deposition and scouring caused a 30% decrease in bed slope and a 40% decrease in bed material size downstream compared to pre-removal conditions. These Impacts are consistent with those of other studies and, hence, are rated a 'strong' linkage. Other 'strong' linkages are reported by the studies of Hill et al. (1993), Bushaw-Newton et al. (2002) or Stanley et al. (2002), all of which revealed a decrease in water temperature upstream, leading to an increase of dissolved oxygen conditions that might favour benthic invertebrate and fish communities.

*Table 6. Qualitative and quantitative evidence for the effectiveness of weir removal and related in-stream modifications.*

| Reference | Type | Qualitative | Quantitative |
|---|---|---|---|
| Kanehl et al. (1997) | Active restoration | After dam removal depth varied considerably following flow variations, rocky bottom increased upstream, bank stability increased upstream and decreased downstream, habitat quality index scores increased dramatically. Short term effects on fish biomass: increase upstream/long term effects: general increase | |
| Bushaw-Newton et al. (2002) | Active restoration | Increased sediment transport has led to major changes in channel form in the former impoundment and downstream reaches leading benthic macroinvertebrate and fish assemblages to shift dramatically from lentic to lotic taxa. No significant upstream–downstream differences in dissolved oxygen, temperature, or most forms of nitrogen (N) and P, were obsvered either before or after dam removal. | |
| Hart et al. (2002) | Review | The overall objectives of this article are to assess the current understanding of ecological responses to dam removal and to develop a new approach for predicting dam removal outcomes based on stressor–response relationships | |
| Pizzuto et al. (2002) | Review | If the impoundment contains relatively little sediment and is significantly wider than equilibrium channels upstream and downstream of the dam, then the primary processes above the dam are likely to be deposition and floodplain construction rather than erosion and incision. Increased sediment supply at the reach scale could destroy alternate bars, pools and riffles, and armored beds. | |
| Shafroth et al. (2002) | Review | Following dam removal, large areas of former reservoir bottom are exposed upstream and may be colonized by riparian plants. Transport of upstream sediment may lead to a pulse of sediment deposition downstream, which combined with increased flooding, may both stress existing vegetation and create sites for colonization and establishment of new vegetation. | |
| Pollard et al. (2004) | Active restoration | Cobble habitat without silt generally supports higher taxonomic diversity than do silted areas | |
| Doyle et al. (2005) | Review | Changes in channel form affect riparian vegetation, fish, macroinvertebrates, mussels, and nutrient dynamics. | |

Many organisms are limited in their recovery by restricted habitat availability, which is considered to be the most important limitation. A recovery of habitat variability required geomorphologic processes similar to pre-damming conditions (Doyle et al. 2005). For fish, two major Impacts can be considered: First, if fish communities are being primarily restricted by the physical barrier itself (i.e. limitation of migration), weir removal is likely to instantaneously restore this Impact. Second, and contrastingly, if fish are being limited by the absence of suitable habitats to complete their life cycle (i.e. habitat for spawning, nursery, foraging), ecological recovery required the re-establishment of pre-removal geomorphologic and hydrologic conditions. If the geomorphologic changes are irreversible, ecological recovery will hardly be possible without controlling quasi-natural geomorphic and hydrological processes.

Another limiting aspect refers to the size of a weir. Orr et al. (2006) concluded that the effect of small dam removal was rather small-scale compared to the natural variability of the entire system (Boulder Creek, USA). This finding suggests that small weir and dam removal measures are not likely to have long-term deleterious effects (see also Thomson et al. 2005).

### 3.5 Rivers – (stretch scale) morphology: remeandering

*Introduction; drivers and pressures*

Since before the industrial revolution, people have intervened and altered the natural condition of streams and rivers. The resultant changes to the aquatic ecosystem are pronounced. Changes in water quantity, quality and the physical structure of the river channel have almost without fail led to changes in the composition of the biotic community inhabiting the river, usually with a reduction in the biological diversity of the aquatic ecosystem.

The scientific basis for the creation or re-instatement of meanders as an effective measure in the rehabilitation of rivers and streams was based upon an international search for all types of literature reporting on the results of the monitoring of this type of intervention.

*Responses*

There are cause effect chains that are referenced from the very left to the very right hand side of the conceptual model. These are, however, often not the result of one piece of research but multiple papers meaning that the chain as a whole was not explored within one project. There are no assurances that the unobserved processes leading to an observed link followed the chain as defined within the conceptual model. From the information collected it was observed that linkages involving changes in morphology such as the development of channel bed features have been given more attention in monitoring projects. This could be due to the ease with which physical features can be assessed and / or the speed with which changes occur in comparison to the biological system. The effect of developing features on habitat variables is often not reported reflecting the less well supported linkages in this part of the model. Authors are often satisfied in describing a feature as encouraging a certain taxon rather than explaining why. Frequently there is a reliance on expert knowledge to make links between variables. It can be seen from the conceptual model that no links were established between passive remeandering and processes following remeandering. This reflects the very low number of projects that undertook passive remeandering as an intervention (6 out of 91 projects). The conceptual models generated as a result of the literature study with all documented response links can be viewed (Appendix 1).

*Organism groups comparison*

Of the BQE elements, macroinvertebrate (MZB) indicators were the most often applied during monitoring exercises (Figure 13). Following macroinvertebrates in descending order of frequency of use are the fish indicator group, the macrophyte indicator group and the phytobenthos indicator group. 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 14). Fish and macrophyte indicator groups 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.

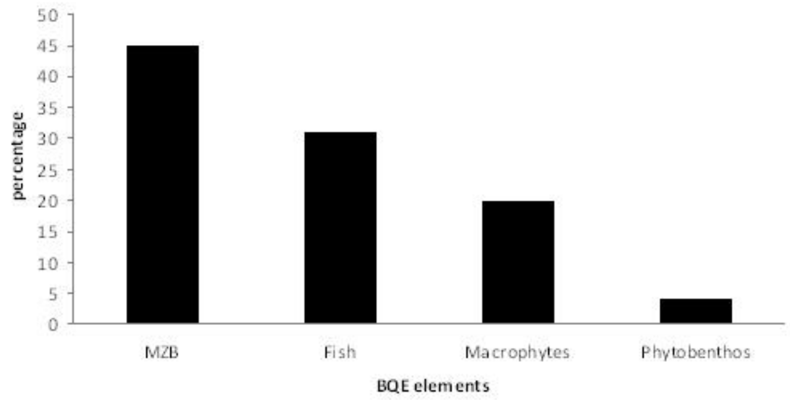



*Figure 13. Percentage representation of BQE elements within monitoring schemes.*

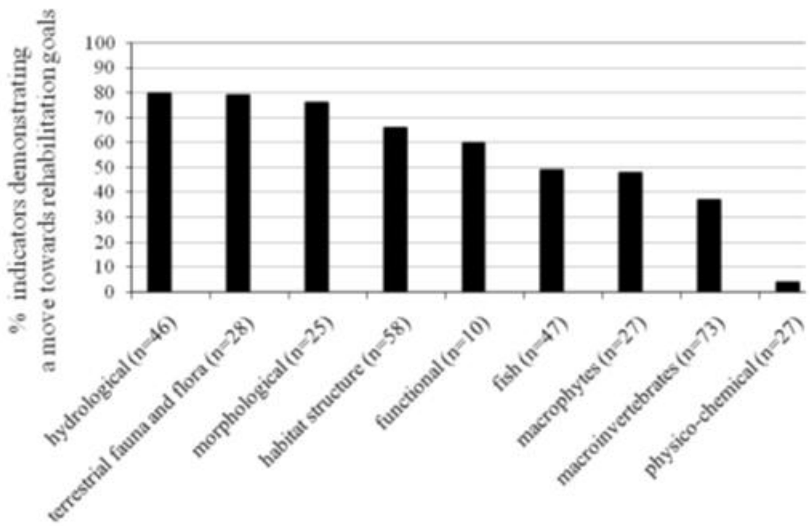



*Figure 14. Positive indicator response per indicator group within the first five years of monitoring. (Matthews et al, 2010)*

*Synthesis*

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’.

Other factors to be kept in mind are that re-meandering was often only one intervention incorporated within a set of interventions applied to a project stretch and it was therefore difficult to establish a direct link between re-meandering and observed changes. This is complicated by a lack of the use of a control in many monitoring programs and a lack of standardization within monitoring. However some general conclusions may be drawn.
External influences that were identified to have influenced the ecological recovery of the project rivers were 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.

Figures 15 and 16 give an overview of indicator responses. The number of equivocal responses refers to projects containing various indicators representing an indicator group that reflect a mix of positive and negative responses to restoration intervention. It is interesting to note that the number of projects exhibiting equivocal responses for ecological indicator groups declined. This may reflect a reduction in variation in indicator response over time.

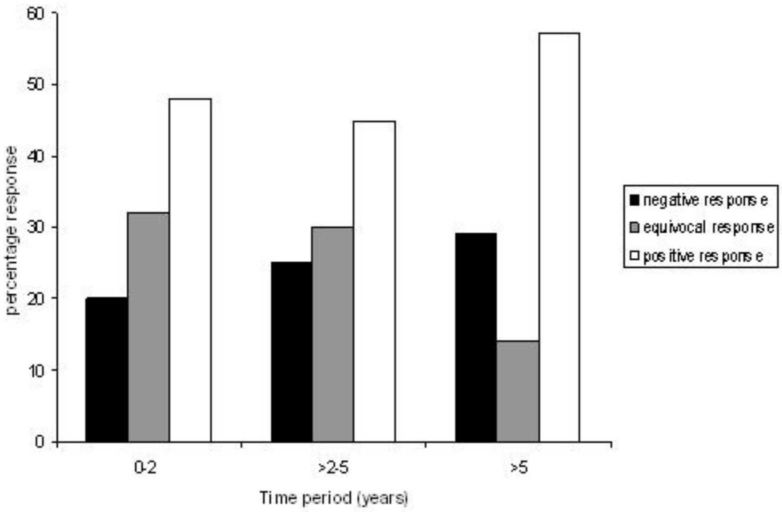



*Figure 15. Ecological indicator response.*

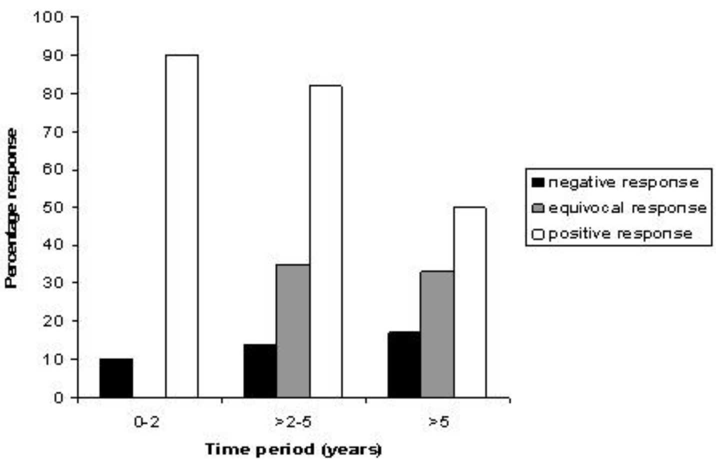


*Figure 16. Non-ecological indicator response*

The monitoring duration set aside for the projects examined are given in Figure 17. If more than one monitoring duration was specified for one project then all monitoring durations were included in the analysis. No distinction was made between delayed and continuous monitoring schemes. Therefore, the graph represents actual time elapsed since the completion of rehabilitation interventions.

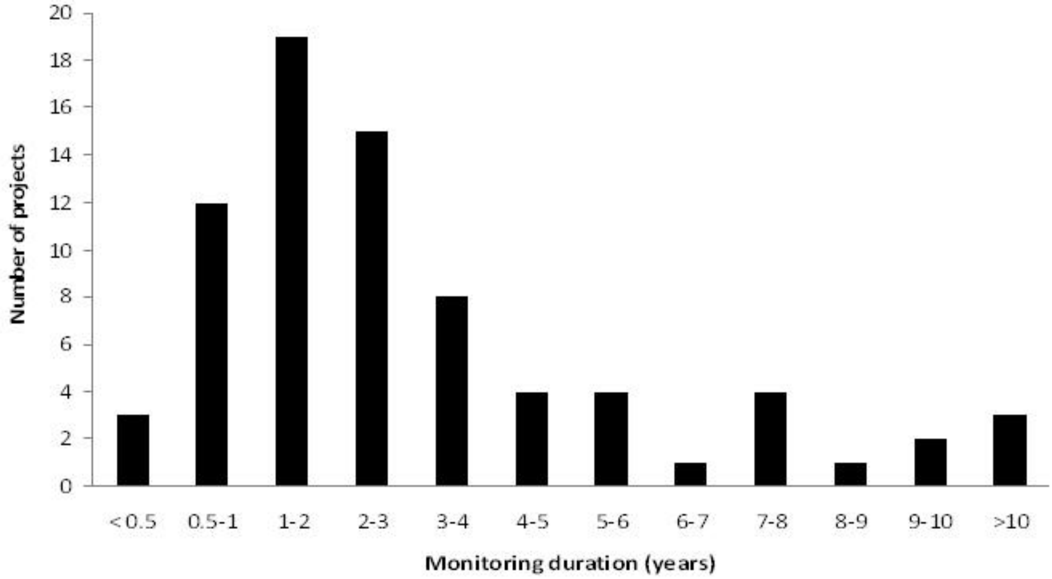



*Figure 17. Project monitoring duration*

The initial recovery period following the implementation of rehabilitation measures and intrinsic disturbance to the system is often characterized by high levels of variance within biological populations. An initial disturbance signal could influence the ability of evaluators to determine the ecological success of a restoration (Tullos et al. 2009). Therefore it makes sense to allow the influence of disturbance to settle prior to measuring the outcomes of rehabilitation. The majority of monitoring examined in the literature survey was delayed for between 1 and 3 years and very projects few delayed monitoring for more than 5 years (Figure 18).

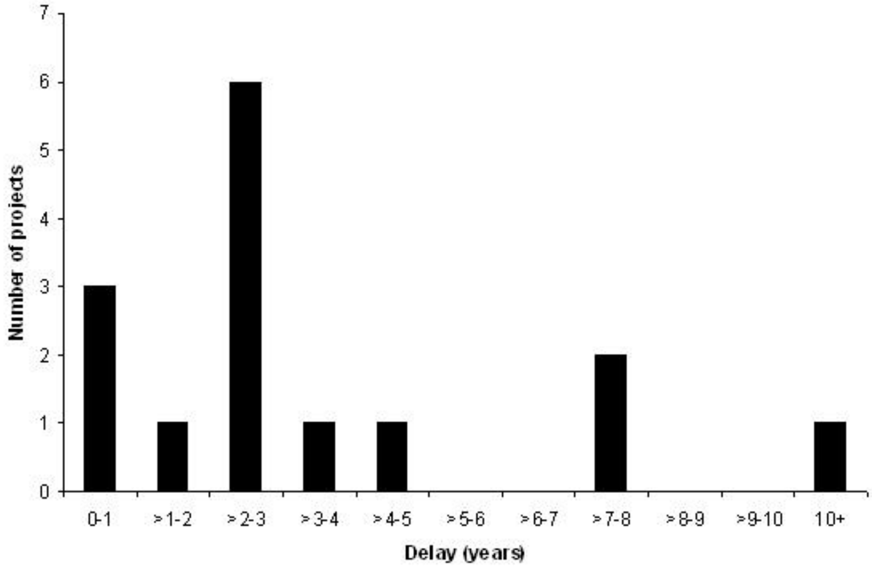



*Figure 18. Delay till commencement of monitoring projects*

Most projects commenced monitoring directly following the completion of restoration measures. Only 15 out of 90 (17%) of projects examined delayed monitoring by any degree following the completion of restoration interventions. Within these 15 projects, 20 monitoring procedures were delayed reflecting differing delays per indicator group e.g. faunal, morphological etc. 17 of the 20 delays were for a period of less than 6 years, two were between 7 and 8 years and one was for 12 years which was a floral survey.

Project size was defined in terms of length of river stretch to be restored (Figure 19). Only five projects were defined in terms of an area (hectares) rather than length which may reflect an inclination for restoration managers to focus on the river channel as apposed to the entire river system. The modal average for restoration size was 1-2 kilometres, 19 out of 67 projects were contained within this category. Another smaller frequency peak was observed at 250 to 500 metres.

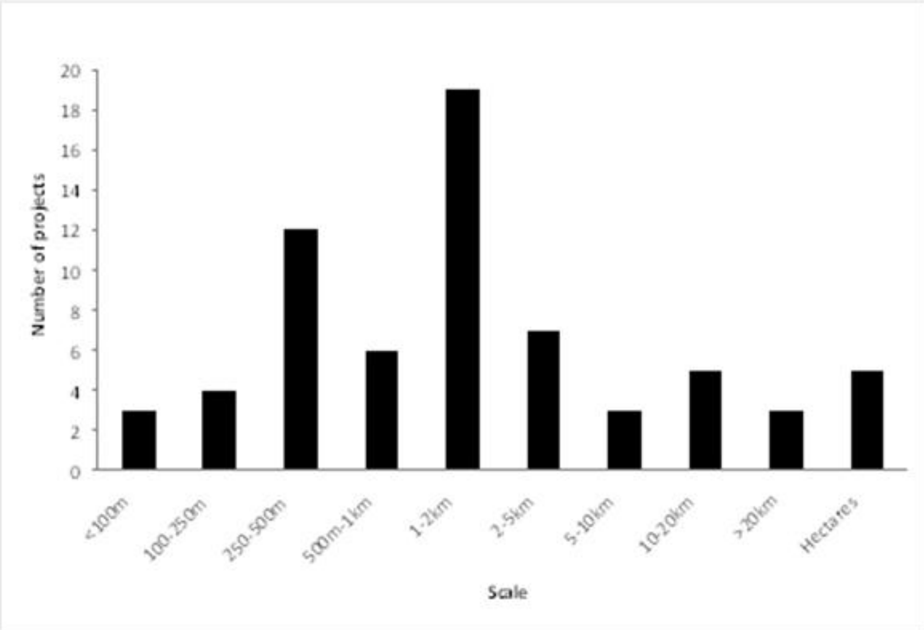



*Figure 19. Scale of restoration projects undertaken*

<u>Recovery time</u>

Judgements were made as to whether an indicator demonstrated a move towards the project aim (reference etc.) or if no aim was defined, the general goals of river restoration. Limited data constrained the analysis, however it can be seen that non-ecological indicator groups (morphology, hydrology) react quickly giving positive indications within the initial few years after project completion. 90% of non-ecological indicators responded positively to restoration intervention within the initial 2 years following restoration. In years 2 to 5 this figure stood at 82%. No trend indicating a particular time period where ecological indicators demonstrated progress could be observed with the available data. Therefore, from these results it could not be established if a particular monitoring year was more likely to yield positive monitoring results in a recovering system. Ecological indicator response demonstrated a similar pattern of response for all years analysed, however this result has to be viewed with reference to the limited data available.

### 3.6 Rivers – (site scale) morphology: instream mesohabitat enhancement

*Introduction: drivers and pressures*

The majority of European river basins and their sub-catchments suffer from such combined impacts on both water quality (organic pollution, eutrophication, toxic compounds) and physical habitat degradation, including simplified habitat structure, barriers to dispersal and biologically unsuitable flow regimes (Friberg 2010a). These stressors—either individually or in combination as multiple stressors—can have severe impacts on aquatic communities of plants and animals and typically cause a loss of biodiversity and ecosystem function. Freshwaters are essential for providing a large array of ecosystem services, in particular provisional services (e.g., drinking water, food) and regulating services (e.g., nutrient spiraling, self-purification, water regulation). The level of degradation that we experience today and the current rate of habitat loss from climate change constitute a serious future challenge if these services are to be maintained.

Whilst there are notable examples of the restoration of water chemistry in river catchments (e.g., Bradley and Ormerod 2002), the dominant paradigm in river restoration has been the rehabilitation of the system, i.e. the manipulation of habitat structure and water flow in order to mitigate adverse environmental and human impacts and ultimately to enhance habitat heterogeneity and biodiversity. These works range from minor reach-scale rehabilitation measures, such as the introduction of gravel bars and large woody debris (LWD), to larger-scale projects aimed at attaining near natural conditions in entire (sub-) catchments (e.g., Hansen and Baattrup-Pedersen 2006; Palmer et al. 2010).

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 removal) of large wood (LWD) provides a key habitat for fish and benthic macroinvertebrates (e.g., Roni and Quinn 2001; Kail et al. 2007) and also stimulates habitat diversity (e.g., creation of pools) by diversifying hydraulic conditions (e.g., Baille 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 review on instream mesohabitat enhancement covered a large number of references on the environmental and biological effects of the introduction of "natural" substrates (large woody debris [LWD], boulders and finer mineral substrata), artificial structures (deflectors) and the removal of artificial bank enforcements (rip-rap). The effects of such structures and its removal, respectively, have been comprehensively reviewed in the recent studies of Roni et al. (2008), Miller et al. (2010) and Palmer et al. (2010), but with a different focus. Roni et al. (2008) primarily addressed fish and biotic production and included measures such as road improvement and management of flood regimes, while the latter two studies investigated the effects of various forms of local and reach-scale habitat improvement (including measures of channel realignment/re-meandering) on benthic invertebrate richness and abundance. In this study, we address cause-effect chains for various animal and plant organism groups and include recovery effects on age structure, biomass, sensitive taxa and functional guilds.

The majority of studies were conducted in North America (60%), while roughly a third of the studies originated from Europe (Figure 20). The vast majority of studies dealt with the introduction of large woody debris (60% of all studies) and boulders (32%); the introduction of spawning gravel and deflector structures were less often applied (12 and 9%, respectively). Detailed information on the study design was provided by almost all references and revealed that restoration and rehabilitation studies on local habitat enhancement are frequently designed prior to the implementation, either following a comparison of monitoring results before and after the measure (BA), a comparison of treated sites (Impact) with untreated control sites (CI), or combinations of both (BACI). Most of the reviewed studies applied the BACI design (44%) followed by those applying a CI design (32%). This is obviously owed to the fairly experimental character of these studies, which often apply statistical analysis including significance testing and, thus, rely on thoroughly designed experiments. On the other hand, however, this may explain a major drawback, as such experiments are most often being designed at relatively small spatial scales (Figure 21) and short time spans (< 5 years between instalment and monitoring, Figure 24). They rarely provide much insight into the large-scale and long-term effects of restoration.

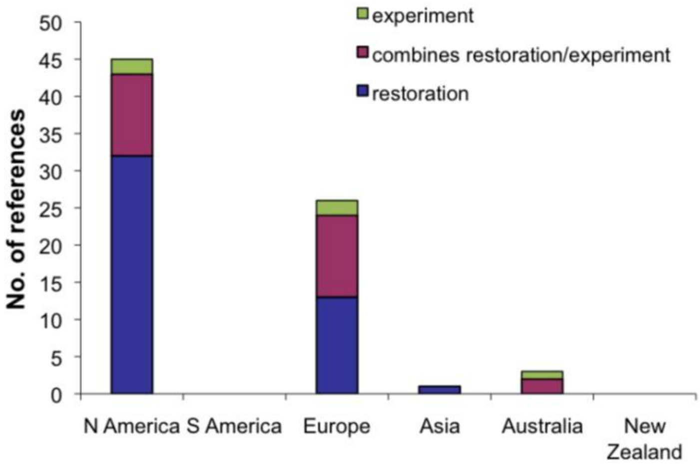



*Figure 20. Origin of N = 75 restoration studies on local in-stream mesohabitat enhancement and removal of bank enforcement structures. The studies are further divided into practical restoration measures, experimental studies and combinations of both.*

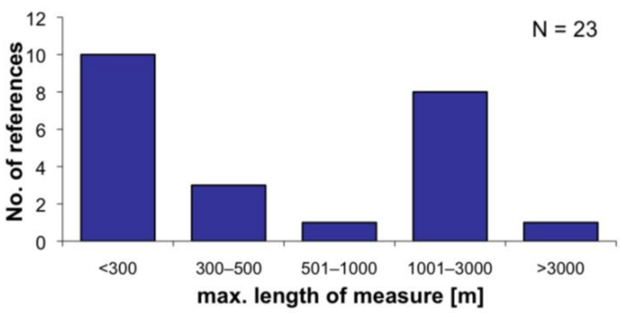



*Figure 21. Lengths of study reaches reported by restoration studies providing the respective information.*

<u>Responses</u>

The introduction of substrates (LWD, boulders, spawning gravel) and deflectors primarily affect three hydromorphological State variables: habitat (substrate) diversity, the provision of cover habitat (for fish) and pool frequency and area (Annex 1.6, Figure 22). Besides the enhancement of habitat diversity and cover through the introduction of (cover) substrates, such as large woody debris, the are effects on pool size and frequency  as many restoration measures aimed at providing pool (winter) habitat and cover for fish (Annex 1.10). Both state variables are frequently reported in restoration studies that aim at recovery of sensitive and economically important fish, such as trout (e.g., Avery 2004; Baldigo et al. 2008) and salmon (e.g., Cederholm et al. 1993; Johnson et al. 2005). The changes in channel morphometry (depth, width) can partly be linked to the increase of pool size and depth, and primarily showed effects on fish. Overall substrate diversity was found to favour both, fish and benthic invertebrate taxa and communities. Substrate diversity is directly enhanced by the addition of LWD (e.g., Harrison et al. 2004; Brooks et al 2006), but also by the placement of mineral substrates such as boulders (Jungwirth et al. 1995; Gerhard and Reich 2000).

Flow heterogeneity, sediment retention and bank stability were also frequently enhanced. Yet, contrastingly, the studies did not provide further information to link these States to biology and, thus, constitute dead ends in our Conceptual Model (Annex 1.6).

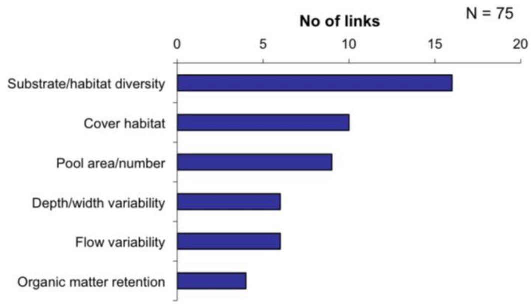



*Figure 22. Ranking of environmental State variables based on the linkages (arrows in Annex 1.6) derived from references on in-stream mesohabitat improvement.*

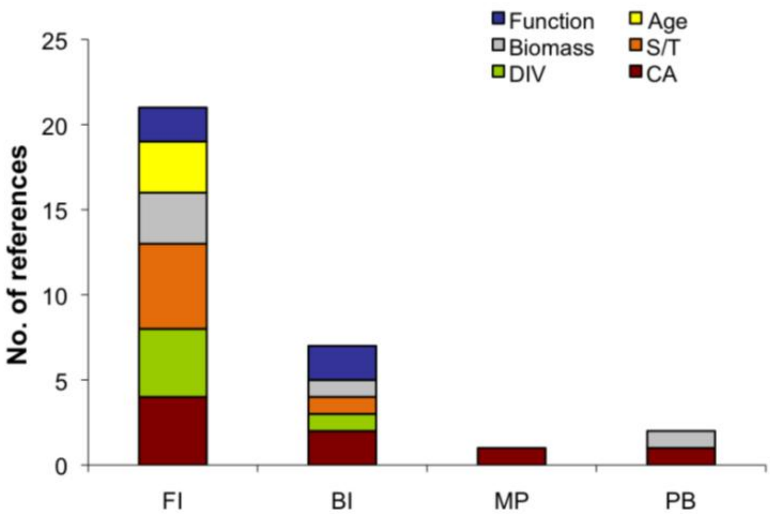



*Figure 23. Number of references addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB). As a study may refer to more than one community attribute, the overall number of references exceeds the number of 75 restoration references reviewed.*

*Organism groups comparison*

By far, most ecological effects were reported for the fish community attributes (21 references) followed by benthic invertebrates (7), macrophytes and phytobenthos (2 each). Figure 23 reveals that several community attributes were equally often referred to in studies on fish and benthic invertebrates. In addition to well-referenced measures of richness and abundance, some studies included effects on sensitive taxa (e.g., on greyling *Thymallus thymallus*: Muhar et al. 2008), age structure (e.g., Jungwirth et al. 1995; Roni et al. 2006), biomass (e.g., Edwards et al. 1984; Coe et al. 2009) and functional guilds (e.g., Larson et al. 2001; Tullos et al. 2006).

Numerous studies consistently concluded that an increased pool-riffle ratio was favourable to the densities of salmonid fish (e.g., Riley and Fausch 1995; Cederholm et al. 1997; Roni et al. 2006). Roni and Quinn (2001) investigated the relationship between LWD density, pool area and the abundance of Coho salmon (*Oncorhynchus kisutch*) during summer. A ten-fold increase of LWD density, for instance, was found to result in a six-fold gain of salmon abundance. However, in parallel a strong decline in the abundance of rainbow trout (*Oncorhynchus mykiss*) along the gradient of increasing pool area was observed. This highlights potential adverse effects of specific in-stream enhancements on different species, even within the same genus.

A remarkable restoration success at the 5th order river Melk in Austria was described by Jungwirth et al. (1995). Enlargements of the cross section, the partial removal of the paved channel bed and bank riprap, and the construction of groynes and bedfalls along a stretch of 1,500 metres yielded clear improvements of the fish fauna. Species richness and diversity were highly related to the morphological parameter of maximum depth variance ($r > 0.85$). A 20-fold increase of this variance resulted in three times more fish species and enhanced Brillouin's diversity by factor 1.5. According to Jungwirth et al. (1995) these strong relationships "… can also be used to forecast the effects of river restoration plans".

Muhar et al. (2008) reported a similar success at the 6th order Alpine river Drau in Austria. The treatments included the removal of riprap, the widening of the riverbed and the initiation of type-specific in-stream structures at the local and reach-scale, resulting in an improvement of up to one ecological quality class (according to the national fish assessment system). The level of enhancement reflected the spatial dimension of the particular rehabilitation and the magnitude of re-established morpho-dynamic processes. The most comprehensive measure (length: 2,000 metres) resulted in a three-fold enlargement of the active channel dimension, and improved the habitat availability for a key fish species, the grayling (*Thymallus thymallus*).

Both Austrian studies demonstrated that fish communities benefit from the enhancement of habitat diversity. Especially the density of juvenile fish can be increased by the creation of shallow areas and gravel bars. Based on the monitoring of various measures aiming at habitat improvement, Binns (2004) observed significant effects on trout abundance. Cederholm et al. (1997) showed that an increase of pool areas positively affected the age structure of Coho salmon (higher density of juveniles) in winter. These findings were similar to those presented by Jungwirth et al. (1995); deep and sheltered areas are preferred winter habitat for juveniles.

The references on invertebrate responses (compare Annex 1.9) to restoration revealed significant effects on the community abundances despite the generally high spatio-temporal variability of this parameter. After LWD treatment, the density of Ephemeroptera preferring pool habitats increased, while abundances of Coleoptera, Trichoptera, Plecoptera and Oligochaeta deceased due to the decreasing proportion of riffle habitats (Hilderbrand et al. 1997). However, LWD and associated habitats were not sampled in this study. Gerhard and Reich (2000) observed highest species richness and abundance on LWD, twigs and CPOM—micro-habitats that only occurred after log placement. The authors could, thus, demonstrate the significant response of the invertebrate fauna to restoration with LWD.

In general, phytobenthos and macrophyte communities have been almost neglected in restoration studies (Figure 23, Annexes 1.7 and 1.8). Only two studies provided evidence for a significant increase of periphyton (phytobenthos) biomass, expressed as Chl-a concentration and/or ash free dry mass (Moerke et al. 2004; Coe et al. 2009). According to the latter publication the increase of habitat surface area after log placements caused the elevated biomass values. Merz et al. (2008) found macrophyte abundance to be favoured by the introduction of spawning gravel into Mokulumne River, USA. The macrophytes, however, covered 70% of the spawning area of chinook salmon (*Oncorhynchus tshawytscha*), which had a significant negative effect on spawning use.

*Qualitative and quantitative links*

Most linkages between restoration measures and biological recovery were described in qualitative terms, while very few studies provided quasi-quantitative relationships (Table 7). According to Hilderbrand et al. (1997) systematic placement of 50 logs at 225 metres channel length increased the pool area by 150% and decreased the riffle area by 40%. The effects exceeded those resulting from random log placement, but nonetheless have to be considered at least constrained to the "low-gradient" river type investigated by the authors. Real quantitative (i.e. quasi-mechanistic) relationships between habitat enhancement and biological recovery are missing in the reviewed body of literature. It is highly questionable if such relationships can be detected at all without sampling a huge population of restored streams, given the high degree of (natural) variability of treated stream reaches, and the multiple stressors that may continue to affect in-stream biota, but whose impact is not detected and not even targeted at all.

Several examples of non-effect studies assumed the absence of biological recovery being 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' controlled and limited restoration effects.

These findings imply that quantified cause-effect relationships should be used with caution for predictive modelling. The results are often based on small numbers of replicates and subjected to specific regional environmental settings, both of which render the transfer of results to other regional settings hardly possible without further testing.

*Table 7. Qualitative and quantitative evidence for the effectiveness of the enhancement of in-stream mesohabitat structures.*

| Publication | Qualitative | Quantitative |
|---|---|---|
| Baldigo et al. (2008) | Unspecified measures (acc. to "Natural Channel Design" approach) led to shift in dominant species and increase of intolerant species richness of the invertebrate community. | |
| Binns (2004) | Trout response to habitat manipulation varied among projects, but acceptable responses occurred across all sizes of streams. Mean increases of wild trout abundance and biomass among different stream orders ranged from 30 to 250%. | |
| Cederholm et al. (1997) | Salmon abundance increased in winter season after treatment by LWD. | |
| Gerhard and Reich (2000) | Increase of invertebrate richness and diversity in sections treated with LWD. | |
| Herbst and Kane (2009) | An increase of EPT taxa richness by 7 taxa followed after the complete relocation/ recreation of 150 m channel. | |
| Hilderbrand et al. (1997) | Systematic placement of 50 logs at 225 m channel length increased the pool area by 150% and decreased the riffle area by 40%, but no significant changes in the macroinvertebrate community were observed. | |
| Jungwirth et al. (1995) | Measures led to increased heterogeneity of water depth and current velocity, and added sandy and muddy in-channel microhabitats. One year after restructuring, the | NFS = 0.00927 * VMD + 6.12, r = 0.86; n = 15; NFS: Number of Fish Species, VMD: Variance of Maximum |

| Publication | Qualitative | Quantitative |
|---|---|---|
| | number of fish species increased from 10 to 19. Fish density and biomass tripled during the period of investigation. The abundance of individual species changed considerably (decrease of *Leuciscus cephalus*, *Gobio gobio*), resulting in a more balanced fish community structure. | Depths<br><br>FSD = 0.0007014 * VMD + 1.28;<br>r = 0.897, n = 15; FSD: Fish Species Diversity |
| Moerke et al. (2004) | Increase in abundance of invertebrates and periphyton in sections treated with LWD. | |
| Muhar et al. (2008) | River bed widening and reconstruction of former side channel at 1900 m river length yield improvement of habitat and fish assessment scores by one quality class. Other restored reaches/sites showed minor improvements. | % aquatic habitat area and fish ecological status highly correlated ($R^2$ = 0.81; n = 6) |
| Riley and Fausch (1995) | Abundance and biomass of adult trout (age-2 and older), and often juveniles (age 1) as well, increased significantly in the treatment sections of each of the six streams after log drop structures were installed. Patterns of change in trout biomass were similar to abundance changes in all streams. | |
| Roni and Quinn (2001) | Juvenile Coho salmon densities were 1.8 and 3.2 times higher in treated reaches compared with reference reaches during summer and winter, respectively. The response of Coho density to LWD placement was correlated with the number of pieces of LWD forming pools during summer and total pool area during winter. | Summer: CDR = 0.59 * LWD - 0.01; $R^2$ = 0.25, n = 27, CDR: Coho salmon density response<br>SDR = -0.83 * PAR + 0.15; $R^2$ = 0.45, n = 20; SDR: age 1+ steelhead trout density response<br>Winter : JDR = 0.25 * PAR + 0.04 ; $R^2$ = 0.27, n = 24 ; JDR: juvenile Coho salmon density response; TFR = -0.42 * PAR + 0.21; $R^2$ = 0.20, n = 20; TFR: trout fry density response |
| Roni et al. (2006) | Both Coho salmon and trout response to boulder weir placement were positively correlated with difference in pool area (p < 0.10), while dace and young-of-year trout response to boulder weir placement were negatively correlated with difference in LWD (p < 0.05). | Pearson's r significant at p < 0.1 (*) and p < 0.05 (**)<br>% pool area/Coho abundance: 0.51*;<br>% pool area/trout abundance (> 100 mm length): 0.54*; LWD/dace: -0.77**; LWD/trout abundance (< 100 mm length): -0.70** |

*Monitoring and recovery time*

The time span between restoration and monitoring of effects was highly variable and ranged from one to 50 years with an average (median) value of 2.5 years (Figure 24). On average, monitoring was performed only twice after restoration and was then compared against before and/or control values (e.g., Gerhard and Reich 2000; Roni et al. 2006; Muhar et al. 2008). Other studies sampled in subsequent years to record the biological succession at the treatment reaches (e.g., Riley and Fausch 1994; Herbst and Kane 2009). Some references reported sampling during various seasons to gain information about seasonal variability of the fish communities (e.g., Jungwirth et al. 1995; Cederholm et al. 1997; Roni and Quinn 2001).

Despite the significant effects documented, the duration and frequency of monitoring is generally low and do not allow of the detection of long-term trends. In this context, Moerke et al. (2004) observed increased habitat evaluation scores after channel modifications, but noted some decline in habitat quality over the following five years due to insufficient sediment trapping upstream. Roni et al. (2008) state that the potential benefits of most instream structures will be short lived (<10 years) unless coupled with riparian planting or other process-based restoration activities supporting long-term recovery of deficient processes. Schmetterling and Pierce (1999) reported adverse effects of a 50-year flood event on pool formation initiated by the introduction of LWD and boulders. The loss of pool habitat was significant in a meandering section due to flood-induced channel migration. Merz et al. (2008) found aquatic macrophytes to immediately cover up to 70% of spawning gravel that was previously introduced to support Chinook salmon (*Oncorhynchus tshawytscha*); spawning use of the section by salmon was significantly reduced.

Synthesis

In summary, the available literature shows two main features: First, the majority of restoration studies on in-stream mesohabitat improvements are short-term and, thus, cannot provide insight into long-term effects. And second, the few references including long-term monitoring results largely imply that habitat enhancement and related biological effects of the most frequent restoration measures (introduction of LWD and boulders) are prone to environmental impacts beyond the scale of restoration. Such impacts include hydrological and geo-morphological processes largely controlled by catchment land use and land cover (Larson et al. 2008; Shields et al. 2008). In consequence, the catchment needs to be considered to render reach-scale restoration successful in the long-term.

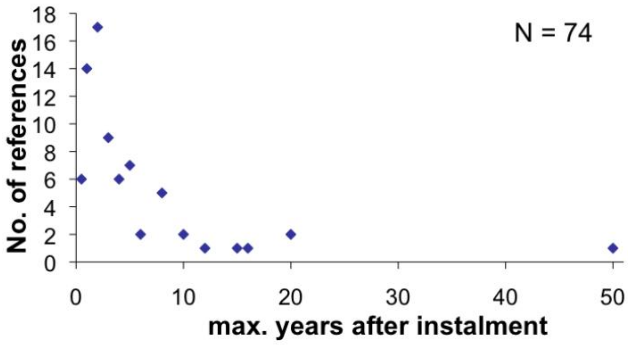



*Figure 24. Time span between restoration and investigation of the relevant reviewed studies.*

Although literature reviews are inevitably biased towards 'positive' findings—failure stories are seldom told and rarely published—a notable finding of this review is that full-success stories are fairly scarce in the body of literature on instream mesohabitat restoration. Irrespective of the many (short-term) positive effects, in particular due to the introduction of LWD, roughly half of the reviewed references imply failures of habitat improvement, biological recovery or both, when it comes to long-term enhancement and recovery. This does not mean that these studies reported total failures, but we simply cannot judge on the level of success of many studies because of their limited monitoring efforts.

The list of potential limiting factors of mesohabitat improvement and biological recovery is comprehensive. Six main aspects are frequently hypothesised: i) inappropriate scaling of restoration, ii) inappropriate timing of monitoring, iii) inappropriate implementation of restoration, iv) inappropriate indicators and indicator groups, v) confounding effects of natural variability and vi) presence of multiple stressors not addressed by restoration.

Palmer et al. (2010) presented an extensive review of (site-scale) measures of habitat improvement (including the placement of LWD and deflectors) on benthic macroinvertebrate communities in the U.S. and Europe and concluded that the overall level of watershed deterioration is relevant to the success of mesohabitat enhancement. Therefore, only the watershed perspectives can provide insight into whether a project will succeed. In addition to the already mentioned abiotic factors acting on the watershed level and affecting restored sites (water quality, sediment input, changes in water temperature) there are also numerous biotic impacts acting on larger spatial and temporal scales. The re-establishment of near-natural assemblages at restored sites requires the colonization by sensitive species. In densely populated, intensively used areas such species might have been brought to extinction in entire catchments, due to long-term pollution and large-scale habitat alteration. The presence of source populations in the catchment and the absence of barriers blocking colonization pathways are crucial for the colonization with sensitive species. Finally, population establishment at restored sites is likely to be ruled by complex competition between already present tolerant species, arriving sensitive species and possibly invasive aliens. These poorly understood biotic interactions currently render prediction of restoration effects on species level almost impossible. Accordingly, Palmer et al. (2010) refer to the hierarchy of actions proposed by Roni et al. (2008): First the critical habitats in the watershed need to be protected; then water quality has to be improved. After that, watershed processes are to be restored (e.g., habitat connectivity, hydrology) and finally the in-stream habitats can be enhanced. Palmer and colleagues advocate the use of 'softer' restoration approaches that do not involve full-scale manipulation of the channel and the riparian zone. In parallel, they call for actions on the larger scale such as storm-water management, changes in forestry or agricultural practices, preservation of land and riparian vegetation to guarantee restoration success. Entrekin et al. (2008) hypothesised that surrounding land use, substrate composition, temperature and the method of log placement are variables that interact and influence the recovery of stream biota to wood additions.

In their comprehensive meta analysis of the effects of mesohabitat enhancement on benthic invertebrates, Miller et al. (2010) blame on the "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 invertebrates 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 used.

Brooks et al. (2002) even argue that high within-study variability and low statistical power may render the use of benthic invertebrates questionable for detecting reach-scale responses to restoration. However, their statement mainly addressed the highly variable abundance patterns of this organism group. In their review of mesohabitat enhancement projects and its effect on fish communities, Roni et al. (2008) concluded that biological effects are highly variable among species, life stages and the type of in-stream structures. Those measures seemed most successful to the authors that create large changes in physical habitat and mimic natural processes. However, restoration effects on the biology were often documented only for comparatively short stream reaches.

### 3.7 Rivers – eutrophication/organic material

*Introduction: drivers and pressures*

Water quality improvement by riparian buffers primarily aims at buffering the adverse impacts of intensive agricultural land use adjacent to streams and rivers. A differentiation between high- and low-energy streams was made a priori and based on the assumption that both natural riparian buffer conditions and typical land uses adjacent to a stream reach differ depending on the stream and floodplain gradients. In both cases, however, 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 (e.g., Barton et al. 1985; Castelle et al. 1994). Riparian trees provide shade and organic material (leaf litter, wood) that have various affects on in-stream biota (e.g., Parkyn et al. 2005; Davies-Colley and Quinn 1998; Davies-Colley et al. 2009).

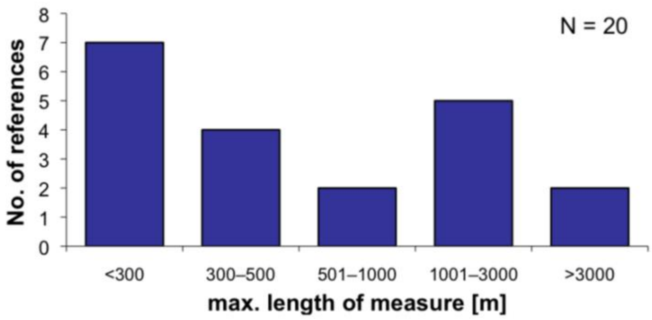



*Figure 25. Lengths of study reaches reported by 20 restoration studies that provide this information.*

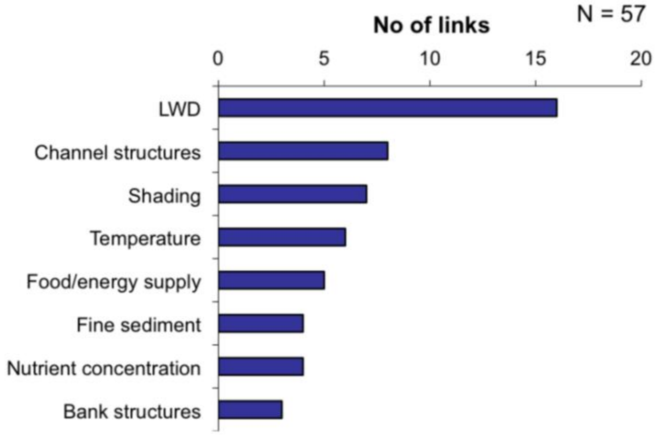



*Figure 26. Ranking of most important environmental State variables based on the linkages (arrows in Annex 1.1) derived from 57 references.*

The restoration of riparian vegetation either refers to active measures, i.e. the instalment of riparian buffers (e.g., Schultz et al. 1995; Northington and Hershey 2006; Sutton et al. 2009) or to passive restoration by allowing riparian buffer strips to establish either with fencing (to exclude large herbivores, e.g., Oppermann and Merenlender 2004) or without fencing (e.g., McBride et al. 2008; Pedraza et al. 2008).

*Responses*

Mixed riparian buffers consisting of trees, shrubs and grass strips are considered to be most effective in the retention of fine sediments and nutrients from both surface runoff and the upper groundwater layer (Correl et al. 2005). Results and suggestions on the minimum width and length of riparian vegetation to effectively buffer fine sediments and nutrients strongly vary. Castelle et al. (1994), for instance, reported a width range of 3–200 m. The authors concluded from their review that a minimum width of 15 m on either side of a stream was sufficient to protect streams under most conditions, while a minimum buffer width of at least 30 m on either side has been found to provide also shading comparable to old-growth riparian forest. Buffers of 30 m width were found to be successful in maintaining macroinvertebrate background levels in Californian streams adjacent to logging activities. A similar minimum width is suggested by Wenger (1999), who in addition developed a function to calculate the minimum buffer width based on the riparian slope (Table 7).
Results on the minimum length of a buffer strip are less frequent. Parkyn et al. (2003) concluded from modelling studies in New Zealand that the minimum length of riparian buffers was 1–5 km for first-order streams versus 10–20 km for fifth-order streams in order to achieve reductions of up to 5° C water temperature. Based on 20 studies that also provided information on the length of the studied sites or reaches (Figure 25), this was less than 1,000 m for 70% of the studies; only two references had study reaches >3 km length.

The complete model reveals fairly complex relationships between the restoration of riparian vegetation, its environmental effects and eventually its impact on in-stream plant and animal assemblages. Based on the frequency with which the linkages are being referred to in the literature, three major chains become obvious through i) enhanced nutrient/sediment retention, ii) increased shading effects and subsequent temperature decrease and iii) increased amounts of large wood (LWD) on the stream bottom. The complexity and interrelation of these state variables is illustrated in Figure 26, based on the number of linkages (arrows) heading to and from the respective State variables.
Consequently, there is evidence that riparian buffer instalment is an effective management measure to increase water quality and habitat complexity (e.g., Dosskey 2001; Broadmeadow and Nisbet 2004; Mankin et al. 2007), to decrease fine sediments and water temperature (e.g., Broadmeadow and Nisbet 2004; Oppermann and Merenlender 2004) and to provide large wood (e.g., Oppermann and Merenlender 2004). Large wood is frequently referred to as a key structure or key habitat that not only provides direct habitat to benthic macroinvertebrates and shelter to young fish (e.g., Brooks et al. 2004), but that also plays a major role in structuring the stream bottom by enhancing the depth and frequency of pools (e.g., Larson et al. 2001; Brooks et al. 2004).

*Organism groups comparison*

There is comparatively little evidence for direct effects of the riparian buffer restoration (Response) on in-stream communities (Impact). A few authors reported an increase of benthic macroinvertebrate richness after riparian restoration (Castelle et al. 1994; Broadmeadow and Nisbet 2004; Pedraza et al. 2008; Becker and Robson 2009; Jowett et al. 2009; Quinn et al. 2009), while their studies do not provide further information on the specific mechanisms causing the observed recovery. The same applies to the study of Penczak (1995), who reported an increase of fish richness and standing crop after passive restoration of riparian trees on the river Warta in Poland.
In contrast, the Conceptual Model revealed a considerable number of relationships between environmental States and biological Impact (Annex 1.3). Altogether, 40 linkages to benthic invertebrate community aspects were reported in 38 restoration studies. The majority of studies (70%)

reported changes in community composition and richness. Five studies only addressed functional aspects of the community. Six studies reported effects on benthic invertebrate biomass and age structure (larval development; e.g., Lester and Boulton 2008; Entrekin et al. 2009). The Conceptual Model revealed four predominant State variables impacting benthic macroinvertebrates: fine sediment, water temperature, food/energy supply (particulate organic matter) and large wood.
Eight references were counted for fish, half of which addressed community composition and richness (compare Annex 1.2). Changes of the fish community were most often related to water temperature and the amount of large wood (LWD). A major effect of large wood is the formation of pools (e.g., Hilderbrand et al. 1997; Chen et al. 2008), which provide a key habitat to young fish (e.g., Cederholm et al. 1997).
Aquatic macrophytes and phytobenthos were less frequently addressed in the literature (compare Annex 1.4 and 1.5, and Figure 27)—although the direct relationships between shading, sediment particle size and nutrient enrichment, and aquatic plants are obvious.

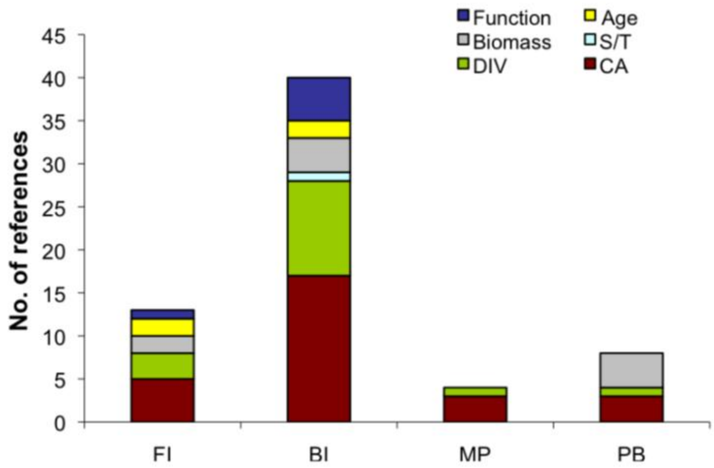



*Figure 27. Number of references addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB). As a study may refer to more than one community attribute, the overall number of references exceeds the number of 38 restoration references reviewed.*

<u>*Synthesis*</u>

There is clear and, in many cases, strong evidence for the role of riparian buffers in controlling nutrient and sediment retention, water temperature and in-stream habitat structure (Table 8). The evidence is based on previous reviews of restoration studies (incl. some strong meta analyses) rather than on single restoration case studies—a finding that is likely owed to the time-scale usually needed for a restoration measure to cause measurable effects, which is rarely covered by the typical time-scale of active restoration projects.
Most of the evidence is of qualitative nature and describes ranges of change of a specific State variable that can be attributed to restoration. Some studies also suggest minimum values for these States that are considered necessary to improve environmental conditions in the long-term.

*Table 8. Qualitative and quantitative evidence for the effectiveness of riparian buffer management and related in-stream habitat improvement.*

| Reference | Type | Qualitative | Quantitative |
|---|---|---|---|
| Dosskey (2001) | Review | buffers can retain pollutants from surface runoff, filter surface and groundwater runoff, stabilize eroding banks, contribute to processes that remove pollutants from stream water flow | |
| Broadmeadow and Nisbet (2004) | Review | removal of riparian woodland can lead to temperature increase up to 4 °C, sufficient LWD and CPOM inputs into the river require buffers of 25–100 m width | |
| Lester and Boulton (2008) | Review (meta analysis) | addition of LWD lead to increase of: fish and macroinvertebrate richness and abundance, macroinvertebrate diversity, bank stability, sediment and organic matter storage, habitat diversity (greater diversity of depths, velocities and habitat elements) | |
| Miller et al (2010) | Review (meta analysis) | addition of LWD and in-stream habitat structure lead to increased macroinvertebrate richness but not density | |
| Oppermann and Merenldender (2004) | Passive restoration | restored reaches had higher frequency of LWD, lower temperature, improved habitat characteristics | |
| Moustgaard-Pedersen et al. (2006) | Active restoration | macrophyte species richness was significantly higher in restored reaches, but plant coverage was not | |
| Castelle et al. (1994) | Review | buffer width of 25–60 m was found to retain 75–95% of fine sediments, buffer widths of 4.5–10 m can retain up to 95% of plant nutrients, buffers of at least 30 m widths have been found to provide shading comparable to old-growth riparian forest and were found to be successful in maintaining macroinvertebrate background levels | |
| Osborne and Kovacic (1993) | Review and active restoration | 10–30 m forested riparian buffer maintain ambient stream temperatures, 9–45 m vegetated buffers retained a substantial portion of sediment in overland, 5–50 m forested riparian buffer retain 60–100% of N and P, riparian forests are more effective in removing nitrate-N from shallow groundwaters than are grass strips | |
| Wenger (1999) | Review | 30 m buffer width sufficient to trap sediments under most circumstances, absolute minimum width is 9 m, 30 m buffers should provide good control of N, 10–30 m native forested riparian buffers should be preserved or restored along all streams to maintain aquatic habitat, protecting diverse terrestrial riparian wildlife communities requires some buffers of up to 100 m width | W = k (s^0,5)<br>W = buffer width<br>k = constant (50 ft)<br>s = slope |
| Warren et al. (2009) | Experiment (no restoration) | volume (V) and frequency (F) of large wood and wood accumulations (wood jams) in streams was most closely associated with the age of the dominant canopy trees in the riparian forest | log10 V ($m^3$/100 m) = (0.0036 * stand age) - 0.2281; $p < 0.001$, $r^2 = 0.80$;<br>F (No./100 m) = (0.1326 * stand age) + 7.3952;<br>$p < 0.001$, $r^2 = 0.63$ |

Nevertheless, there is a clear lack of evidence for strong relationships to the aquatic biota. Only two studies reported effects on benthic invertebrate richness (Miller et al. 2010) and aquatic macrophyte richness (Moustgaard-Pedersen et al. 2006), while other organism groups and community attributes remain unaddressed.

The majority of references report qualitative results. This is sufficient to define minimum requirements for effective (and maybe also successful) restoration of riparian buffers, but such studies rarely provide the statistical relationships and mechanisms needed to predict the effects of management. Only two studies (Wenger 1999; Warren et al. 2009) include quantifiable results and even provide formulas that might be used for predictive modelling in other studies. However, the general applicability of these results in other regions or on other continents is unclear and would require testing.

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 modelling the effects of restoration and biological recovery.

*Recovery time*

Based on some theoretical considerations 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 large wood and, hence, to gain the desired effect on the in-stream hydromorphology (bed form processes and habitat improvement).

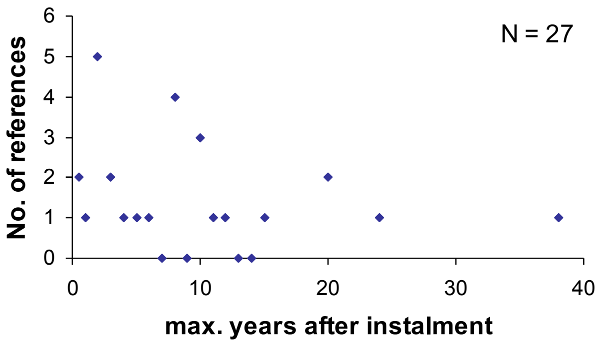



*Figure 28. Time span between restoration and investigation of the relevant reviewed studies.*

Twenty-seven references provided information on the timing of field surveys or monitoring relative to the instalment of restoration measures (Figure 28), two thirds of which were conducted less than 10 years after instalment. This time-scale may be sufficient to detect the effects of direct 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.

Only three out of 38 studies revealed no (or almost no) effects with regard to the anticipated effects hypothesised a priori. Larson et al. (2001) expected positive effects of the addition of large wood to six streams in Washington, North America. Although the frequency of pools increased in all streams, there were no effects detectable for the benthic invertebrate communities up to ten years after wood addition. The authors attributed their negative findings to watershed-scale disturbances, in particular to increased loads of fine sediments.

Sutton et al. (2009) investigated the effects of active riparian buffer restoration on nutrient retention up to eight years after planting streamside vegetation (trees and managed grassland). The authors observed constant nutrient concentrations although the mean forested buffer density in the 15 stream reaches increased from 33 to 44%. This was attributed to insufficient buffer age, width and to gaps in the buffer largely reducing effectiveness). Overall, the buffer restoration was considered not extensive enough to have measurable effects on stream water quality.

Becker and Robson (2009) investigated the effects of riparian buffer re-vegetation on in-stream benthic invertebrate communities in Southern Victoria, Australia. Non-native willows had been removed and replaced by native tree species. Up to eight years after re-vegetation, there was no measurable effect on the benthic invertebrate community. The authors assumed re-vegetation of buffers requiring much more time for positive effects on the biota.

Another study showed strong effects, but implied limited transferability of results to other regions. Warren et al. (2009) quantified large wood loading to 28 streams in the north-eastern United States covering a wide range of in-stream and riparian forest characteristics. They document the current volume and frequency of occurrence of large wood in streams with riparian forests varying in their stage of stand development as well as stream size and gradient. The authors developed regression models to predict the amount and volume of large wood using riparian forest age as descriptor (compare Table 7). The application of their models to other regions, however, revealed that the regression models couldn't be transferred because of regionally different forest characteristics and the legacy of forest land use.

## 3.8 Estuarine and coastal marine waters – multiple stressors

*Introduction: drivers and pressures*

Many estuarine and coastal marine ecosystems have increasingly experienced degradation caused by multiple stressors. Anthropogenic pressures alter natural ecosystems and the ecosystems are not considered to have recovered unless secondary succession has returned the ecosystem to the pre-existing condition or state. Multiple stressors include hydromorphological and sediment barriers (e.g. dams), toxic chemical pollutants, excess nutrient inputs, hypoxia, turbidity, suspended sediments, and other ecosystem alterations, which can impact resources through single, cumulative or synergistic processes. Ecosystem degradation and pollution problems are correlated with increases in population density.

Anthropogenic pressures altering natural ecosystems are largely the result of societal and economic development (Borja and Dauer 2008). Natural ecosystems may recover from anthropogenic perturbations when secondary succession returns the ecosystem to the pre-existing condition or state. However, depending upon the scales of time, space and intensity of anthropogenic disturbance, return along the historic trajectory of the ecosystem may: (i) follow natural restoration though secondary succession; (ii) be re-directed through ecological restoration, or (iii) be unattainable.

*Responses*

Six recovery mechanisms associated to different stressors were identified: (i) recovery from sediment modification, which includes all aspects of dredging and disposal; (ii) recovery by complete removal of stressors limiting natural ecosystem processes, which includes tidal marsh and inundation restoration; (iii) recovery by speed of organic degradation, which includes oil discharge, fish farm wastes, sewage disposal, and paper mill waste; (iv) recovery from persistent pollutants, which includes chemical discharges, such as TBT; (v) recovery from excessive biological removal, related to fisheries; and, (vi) recovery from hydrological and morphological modification.

*Recovery time*

According to Borja et al. (2010), the time-span of recovery after removal of the pressure is highly variable (Table 9), extending from several months (in the case of meiofauna) to more than 22 years (in hard bottom macroalgae and some seagrass species). 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 1.5-10 years for recovery, although some sensitive organisms (such as angiosperms) may take over 20 years to recover. 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 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.

*Table 9. Summary of time for recovery, for different biological elements and substrata, under different pressures. Modified from Borja et al. (2010).*

| Biological elements | Pressure | Intertidal/subtidal | Time for recovery |
|---|---|---|---|

| | | | |
|---|---|---|---|
| **Benthic invertebrates** | **Eutrophication/Pollution by organic matter** | | |
| | Oxygen depletion | Subtidal | 2 yr |
| | Wastewater discharge | Subtidal | 7-20 yr |
| | Sewage sludge disposal | Subtidal | 3->14 yr |
| | Mine tailings | Subtidal | 4->15 yr |
| | Fish farm | Subtidal | 2->7 yr |
| | Pulp mill | Subtidal | 6-8 yr |
| | Eutrophication | Subtidal | >3->6 yr |
| | TBT | Subtidal | 3-5 yr |
| | Oil-refinery discharge | Intertidal/Subtidal | 2-3 yr |
| | **Hydrological-morphological modification** | | |
| | Land claim | Intertidal | 2 yr |
| | Sediment disposal | Intertidal | 3-18 months |
| | Sediment disposal | Subtidal | >5 yr |
| | Dyke and marina construction | Intertidal/Subtidal | 2-3 yr |
| | Aggregate dredging | Subtidal | 2-4 yr |
| | Dredging | Intertidal/subtidal | 2->5 yr |
| | Realignment of coastal defences | Intertidal | >6 yr |
| | Physical disturbance | Intertidal/Deep-sea | 3->7 yr |
| | Fish trawling | Subtidal | 2.5-10 yr |
| | Lagoon isolation | Subtidal | >3 yr |
| | Aggregate dredging | Subtidal | 2-4 yr |
| **Fishes** | **Eutrophication/Pollution by organic matter** | | |
| | Wastewater discharge | Subtidal | 3-10 yr |
| | Oil-refinery discharge | Intertidal/Subtidal | 2-3 yr |
| | **Hydrological-morphological modification** | | |
| | Marsh restoration | Subtidal | 1-2 yr |
| | Dyke and marina construction | Intertidal/Subtidal | 2-3 yr |
| | Marsh and tidal restoration | Intertidal/subtidal | 5-20 yr |
| | Dredging | Intertidal/subtidal | 2->5 yr |
| | Sediment disposal | Subtidal | >5 yr |
| **Macroalgae & seagrasses** | **Eutrophication/Pollution by organic matter** | | |
| | Wastewater discharge | Intertidal | >6->22 yr |
| | Wastewater discharge | Subtidal | 7-20 yr |
| | **Hydrological-morphological modification** | | |
| | Land claim | Subtidal | >20 yr |
| | Dredging | Intertidal/subtidal | 2->5 yr |

| | | |
|---|---|---|
| Sediment disposal | Subtidal | >5 yr |
| Realignment of coastal defences | Intertidal | >6 yr |
| Marsh and tidal restoration | Intertidal/subtidal | 5-20 r |

## 4. Discussion and conclusions

This deliverable aims to identify and compare the differences between cause-effect-recovery chains of different drivers within water categories and between organism groups. For each of the water categories we surveyed literature on information to draw up cause-effect-recovery chains. To identify and compare the different causes, effects and recovery changes, either structural and functional parameters or (meta)data analysis were searched for.
The hypotheses are:

- *At a catchment scale, we anticipate that nutrient stress will be more important in lakes and estuarine and coastal marine systems, while hydromorphological stress will be more important in rivers.*
- *The underlying effects of nutrients will be related to functional production/decomposition processes.*
- *The alterations in hydromorphology will affect habitat availability in rivers.*

<u>*Drivers and pressures in all water categories*</u>
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).
There is a common agreement drivers and pressures in general are the same in lakes, rivers and estuarine and marine 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.

<u>*Responses to recovery measures*</u>
In lakes most studies dealt with eutrophication reduction measures, either source or effect related, to decrease phosphorus loads (less to nitrogen). The responses of organism groups were studied within the food web relations or cascades. An overall overview of responses is given (Table 10).

*Table 10. Lakes-eutrophication with stressor total phosphorus.*

| organism group | recovery driver | responses in structure | functional responses | recovery time | recovery barrieres |
|---|---|---|---|---|---|
| phytoplankton | P | seasonal shifts in major groups | decrease in production | 5-25 yrs | sediment P, N, Si, climate change, fishery management |

| | | | | | |
|---|---|---|---|---|---|
| macrophytes | clarity | depth, richness, indicators | | | physical distribution, loss seed bank |
| zooplankton | plankton food quality, macrophyte structure = release predation pressure | diversity and abundance increase | biomass increase | (1-3):5-25 yrs | sediment P, fishery management |
| macro-invertebrates | tP, delivery organic detritus to sediment, improvement O2, release predation pressure | increase richness, diversity, decrease abundan ce, deep water colonisation, decrease CHIR:OLIG | | 5-25 yrs | sediment P, biomanipulation |
| fish | | decrease biomass, increase piscivorous and percid fish sp | | | fish management |
| waterfowl | | dependent on inverts or phytes | | | interactions macrofauna, macrophytes |

The lake and stream acidification studies mainly concerned liming. Liming is an effect related measure that has to be repeated in time to sort effect. Responses were related to indicators of the reference and diversity. An overall overview of responses is given (Table 11A and 11B).

*Table 11. A: Lakes-acidification with stressor sulphate.*

| **organism group** | **recovery driver** | **responses in structure** | **functional responses** | **recovery time** | **recovery barrieres** |
|---|---|---|---|---|---|
| phytoplankton/ fish | rise in pH by liming | deviations, return acid sensitive spp., increase diversity | uncorreleted to structure, altered food web | ? | climatic variability, droughts, poor habitat quality, limited connectivity, dispersal abilities, biological interactions, invasion of pest species, priority effect, |
| macro-invertebrates/ zooplankton | | increase species diversity and biomass | | ? | |

*B: Lakes & streams-acidification with stressor sulphate.*

| **organism group** | **recovery driver** | **responses in structure** | **functional responses** | **recovery time** | **recovery barrieres** |
|---|---|---|---|---|---|
| zoo-plankton | rise in pH by liming or natural rise in pH | increase, incomplete approximation to communities in reference lakes | lack of predators keeps abundances high | unknown, and potentially confounded by broad-scale environmental change. Spatially synchronous dynamics in acidified and reference lakes | community closure, blocking re-invasion (priority effect), recurrent re-acidification events, landuse change, increasing water colour |
| macro-invertebra tes | natural rise in pH | increase acid sensitive spp., incomplete approximation to communities in reference lakes | acid tolerant herbivores and shredders have replaced lost grazers, lack of predators keeps abundances high | unknown, and potentially confounded by broad-scale environmental change. Spatially synchronous dynamics in acidified and reference lakes | |

Stream restoration was studied for weir and dam removal, remeandering, instream habitat enhancement and re-introduction of riparian buffers. As to be expected weir and dam removal improved connectivity for fauna, all other measure focus on habitat improvement. The response was always expressed in species composition and diversity. An overall overview of responses is given (Table 12A, 12B, 12C and 12D).

*Table 12. A: Streams – hydrology: removal of weirs and dams.*

| organism group | recovery driver | responses in structure | functional responses | recovery time | recovery barrieres |
|---|---|---|---|---|---|
| macroinvertebrates | connectivity, flow and sediment diversity, decrease upstream water temperature and increase in oxygen | | | | up- and downsteam unsuited habitat conditioons |
| fish | connectivity | | | | |

*B: Streams – (stretch scale) morphology: remeandering with stressor habitat loss.*

| organism group | recovery driver | responses in structure | functional responses | recovery time | recovery barriers |
|---|---|---|---|---|---|
| macroinvertebrates | | slow | | | absence of upstream source population, upstream management practices, nutrient enrichment, hydrological change and associated sedimentation/erosion, project size |

*C: Streams - (site scale) morphology: instream mesohabitat enhancement with stressor microhabitat loss.*

| organism group | recovery driver | responses in structure | functional responses | recovery time | recovery barrieres |
|---|---|---|---|---|---|
| macroinvertebrates | habitat structures | increases of EPH, decreases of Col, Tri, Ple, Oli; increases in richness, abundance | | | water quality problems, fine sediment entries, inappropriate scaling, inappropriate implementation, confounding effects of natural variability, presence of multiple stressors not addressed by restoration, colonisation barriers/connectivity problems, invasise aliens, priority effect |
| fish | habitat structures | in- and decreases in diversity | | | |
| macrophytes | gravel | increase cover % | | | |
| phytobenthos | surface increase | Chla increase | biomass increase | | |

*D: Streams – eutrophication/organic material: improvement by riparian buffers.*

| organism group | recovery driver | responses in structure | functional responses | recovery time | recovery barrieres |
|---|---|---|---|---|---|
| macroinvertebrates | fine sediment, water | increase richness, change in | change in biomaas and age | | |

| | | | | | |
|---|---|---|---|---|---|
| | temperature, food/energy supply, large wood | composition | structure | | |
| fish | temperature, large wood | increase richness | | | |
| macrophytes | shading, sediment particle size, nutrient enrichment | effect on richness | | | |
| phytobenthos | shading, sediment particle size, nutrient enrichment | | | | |

Estuarine and coastal marine restoration proejcts were scarce. Studies strongly differed in type of stressor. The response was always expressed in biomass, indicators, species composition, richness and diversity. An overall overview of responses is given (Table 13).

*Table 13. Estuarine and coastal marine waters – multiple stressors.*

| organism group | | Recovery driver | Responses in structure | Functional responses | Recovery time | Recovery barriers |
|---|---|---|---|---|---|---|
| **Benthic invertebrates** | | removal of oil refinery, removal or movement of outfall, sewage-sludge disposal cessation, decrease wastewater discharge, removal farm fish (decrease organic enrichment), increase oxigen | increase richness, diversity, long-live species, sensitive species and coverage area; decrease oportunistic species | Change to species with same functional role as in reference sites; increase biomass and abundance | 2-20 years | Organic enrichment, nutrients in sediment. Dispersal of reproductive stages due to storms. Recovery dependent of constituent species, which have different life-cycles, reproduction periods and patterns of larval dispersal |
| | hydro-morphological pressures | realigment of coastal defenses, drecrease or stop dredging, sediment dumping and harbour construction | return to species characteristic; increase richness and diversity; decrease oportunistic species | | from months to 10 years | |
| **Macrophytes** | | decrease wastewater discharge, decrease N loading | changes in species; increase coverage | | up to 22 years | nutrients in sediment, long residence times, watershed management, internal loads due to sediment release |
| | hydro-morphological pressures | sediment disposal, land claim, realigment of coastal defenses | increase coverage and biomass; decrease *Phragmites* or *Typha* and reestablishment of tidal salt marsh vegetation (in marshes) | increase annual leaf production, and C and N incorporation and consequently increase water quality | 2-10 years | Reclamation for port and industrial complex construction (increasing water turbidity and water velocity), heavy boat traffic, comercial oyster culture (increase in sedimentation and direct physical disturbance),. Salinity, hydroperiod (through its influence on soil redox potential and sulfide accumulation). Dredged materials that can be redistributed by wind-induced waves and currents |

| | | | | | | |
|---|---|---|---|---|---|---|
| **Fishes** | | decrease wastewater discharge (increase oxigen), decrease fish trawling, removal oil refinery | increase richness, increase rare and vulnerable species | | 2-10 years | organic enrichment, nutrients in sediment, previous recovery of benthic communities on which the fishes feed |
| | hydro-morphological pressures | dike and marina construction, realigment of coastal defenses, mash and tidal restoration | increase richness and abundance; return to reference fish assemblages | increase feeding, growth and survival | 1-20 years | |

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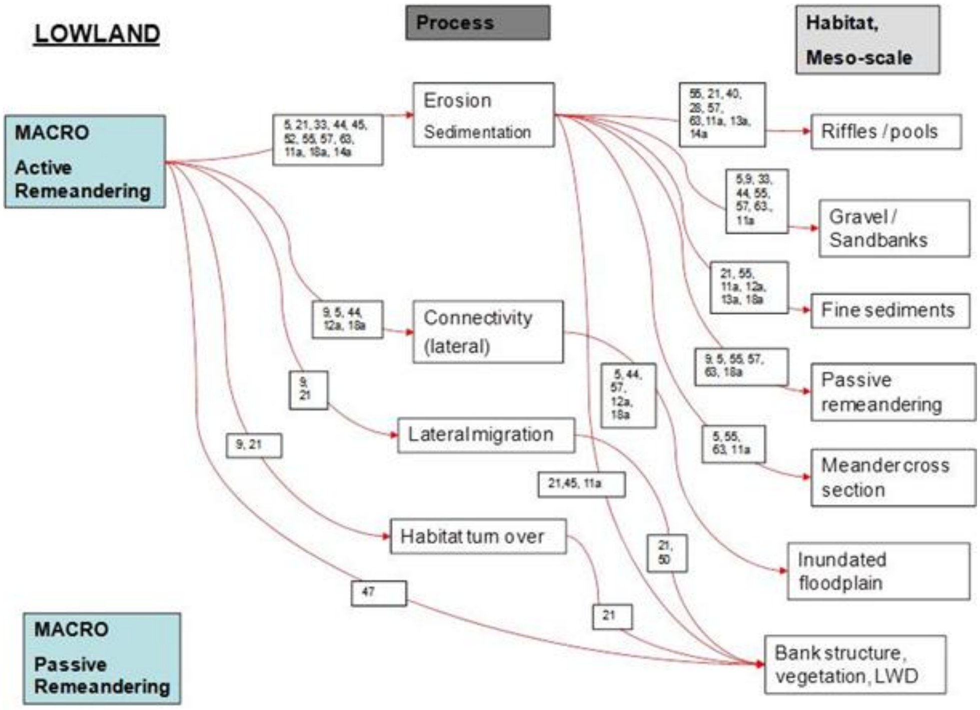
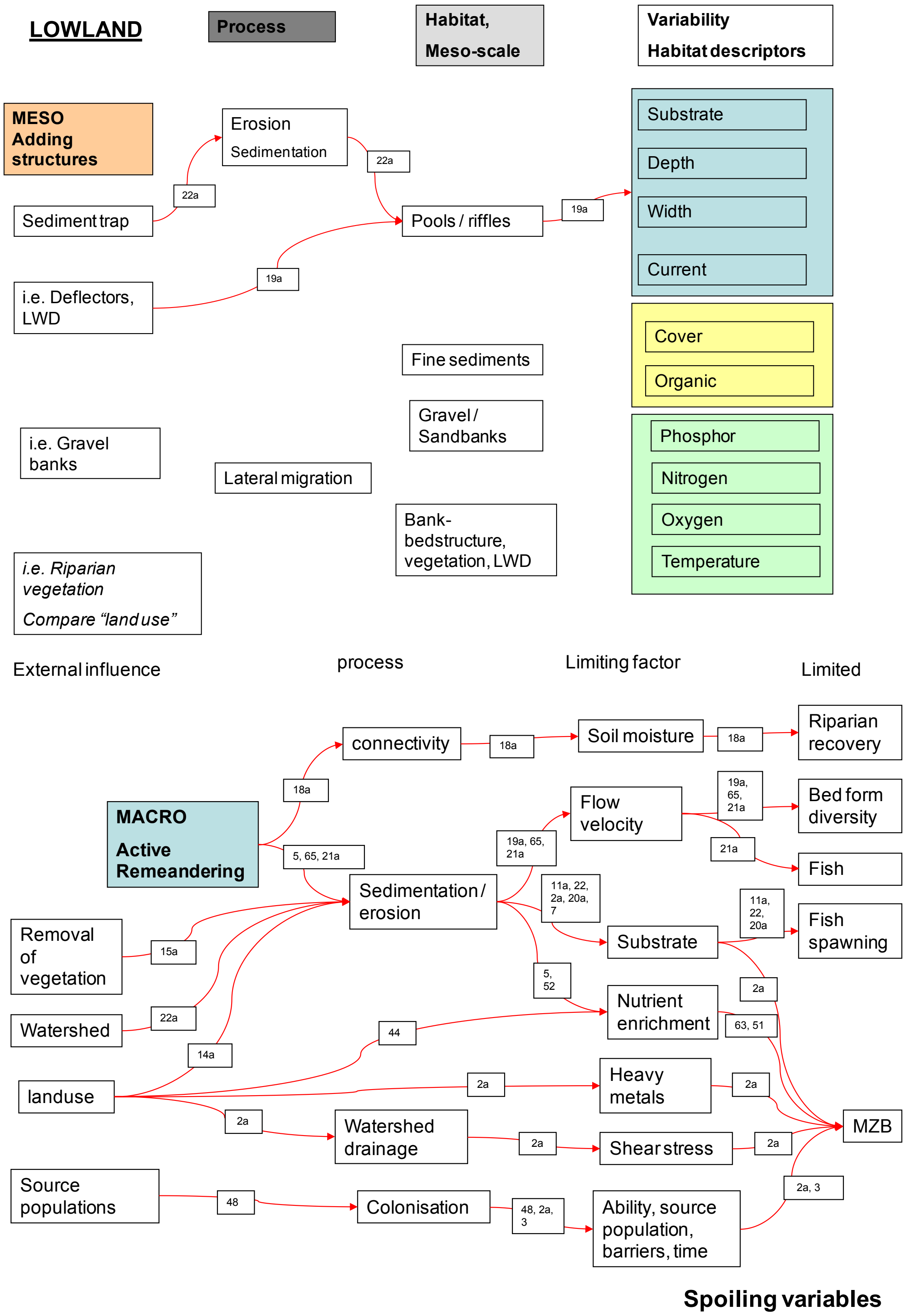
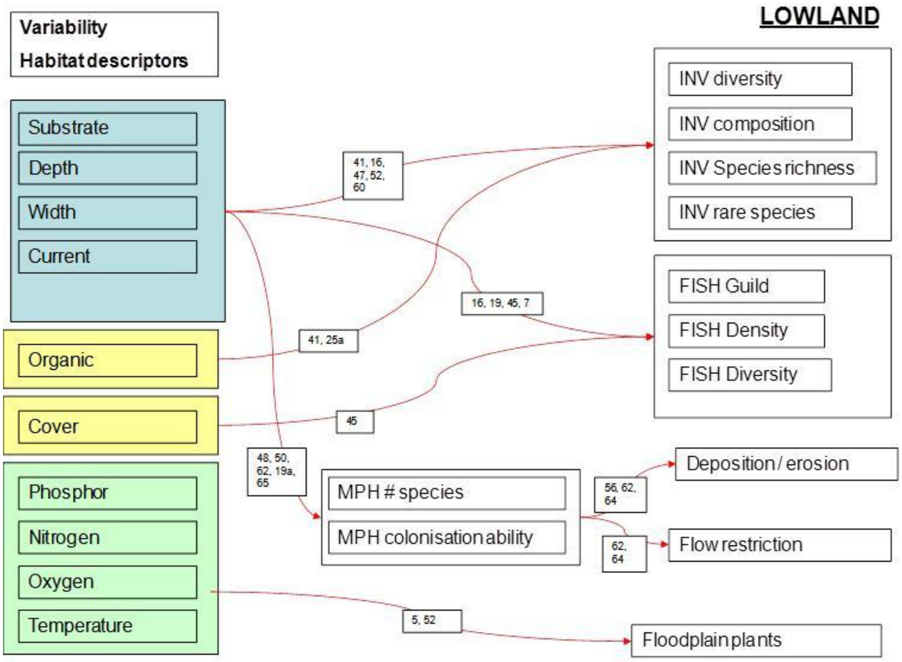
**Appendix 1**

Conceptual models of cause effect chains following remeandering of rivers and streams

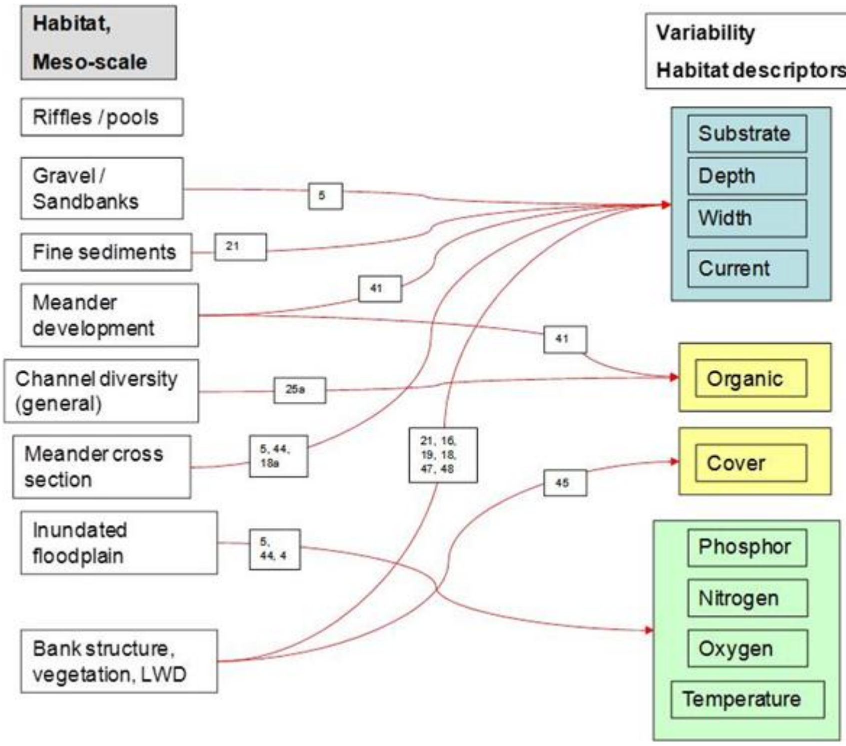
Note: Numbers refer to references that can be viewed in the accompanying excel document. Numbers followed by the letter ‘a’ indicate references where remeandering occurred at locations that could not be confirmed as lowland rivers or streams.




LOWLAND
Process
Habitat,
Meso-scale
Variability
Habitat descriptors
MESO
Adding
structures
Erosion
Sedimentation
22a
22a
Sediment trap
Pools / riffles
19a
Substrate
Depth
Width
Current
i.e. Deflectors,
LWD
19a
Cover
Organic
Fine sediments
Gravel /
Sandbanks
Phosphor
Nitrogen
Oxygen
Temperature
i.e. Gravel
banks
Lateral migration
Bank-
bedstructure,
vegetation, LWD
i.e. Riparian
vegetation
Compare "land use"
External influence
process
Limiting factor
Limited
connectivity
18a
Soil moisture
18a
Riparian
recovery
18a
MACRO
Active
Remeandering
Flow
velocity
19a,
65,
21a
Bed form
diversity
5, 65, 21a
19a, 65,
21a
21a
Fish
Sedimentation /
erosion
11a, 22,
2a, 20a,
7
11a,
22,
20a
Fish
spawning
Removal
of
vegetation
15a
Substrate
5,
52
2a
Watershed
22a
Nutrient
enrichment
63, 51
44
14a
Heavy
metals
landuse
2a
2a
2a
MZB
Watershed
drainage
2a
Shear stress
2a
2a, 3
Source
populations
48
Colonisation
48, 2a,
3
Ability, source
population,
barriers, time
Spoiling variables



LOWLAND
Variability
Habitat descriptors
Substrate
Depth
Width
Current
Organic
Cover
Phosphor
Nitrogen
Oxygen
Temperature
41, 16, 47, 52, 60
16, 19, 45, 7
41, 25a
45
48, 50, 62, 19a, 65
5, 52
INV diversity
INV composition
INV Species richness
INV rare species
FISH Guild
FISH Density
FISH Diversity
MPH # species
MPH colonisation ability
56, 62, 64
62, 64
Deposition / erosion
Flow restriction
Floodplain plants

Habitat,
Meso-scale
Riffles / pools
Gravel / Sandbanks
Fine sediments
Meander development
Channel diversity (general)
Meander cross section
Inundated floodplain
Bank structure, vegetation, LWD
5
21
41
41
25a
5, 44, 18a
21, 16, 19, 18, 47, 48
45
5, 44, 4
Variability
Habitat descriptors
Substrate
Depth
Width
Current
Organic
Cover
Phosphor
Nitrogen
Oxygen
Temperature