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WISER

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

DELIVERABLE

Deliverable D5.1-1: Conceptual Models and effects of river rehabilitation and restoration measures on aquatic organisms

Lead contractor: **University of Duisburg-Essen (UDE)**

Contributors: **Christian K. Feld (UDE), Sebastian Birk (UDE), Daniel Hering (UDE), Anahita Marzin (CEMAGREF), Andreas Melcher (BOKU), Dirk Nemitz (UDE), Florian Pletterbauer (BOKU) and Didier Pont (CEMAGREF)**

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Content

Non-technical summary

Anthropogenic degradation of aquatic ecosystems—rivers, lakes, estuaries and coastal waters—is manifold, pervasive and dates back for centuries in Europe. The ecosystems are affected by physical, chemical, hydrological and morphological modifications, all of which impose environmental pressures on the structure and function of aquatic communities. Human impacts on aquatic ecology have frequently been studies and numerous indicators for assessment and monitoring of various environmental impacts on aquatic ecosystems were developed.

In response, the knowledge about the linkages between environmental pressures and aquatic communities was used to derive appropriate measures to rehabilitate and restore aquatic ecosystems. Restoration ecology is often assuming that communities are beginning to recover as soon as the pressures are reduced or removed. However, the simple reversal of degradation equally often does not show the desired and anticipated ecological effect and the biota continue to stay ‘degraded’. Firstly, the small spatial scale of many restoration measures does not fit the often very broad-scale degradation at the catchment level; secondly monitoring activities are rather short-term and do not sufficiently account for long time periods required for restoration; and thirdly, the knowledge about a catchment’s potential for recovery is sparse.

This report on Conceptual Modelling of restoration and recovery presents a framework to summarize and structure the current knowledge on the effectiveness of river restoration measures and its impacts on the in-stream plant and animal communities. The Conceptual Models are used to illustrate the relationships between three common restoration measures, its effects on instream environmental key variables and the impact of the changing variables on benthic algae, macrophytes, benthic invertebrates and fish. The restoration measures are: i) instalment of riparian buffers to improve water and habitat quality, ii) placement of in-stream structures to improve the mesohabitat and iii) removal of weirs to restore connectivity, hydrology and geomorphology.

The Conceptual Models are used to identify well-known cause-effect chains for restoration, but also knowledge gaps based on the peer-reviewed scientific literature and amended by selected reports from the grey literature.

Terms and definitions

- *Adaptive (ecosystem) management*: paradigm for river basin (restoration) management that is adaptive in its behaviour. Adaptive ecosystem management is an iterative, stepwise approach that involves synthesis of available information in an ecosystem context to define the problem, public participation in goal setting (e.g. protection and restoration of native biodiversity), research and peer review to define science-based management actions (e.g., re-regulation), effective monitoring and evaluation of management actions and adaptive revision of actions based on new information from scientific research (Stanford et al. 1996).
- *Conceptual Model*: a map of entities (concepts) and their relationships. Here, this map is used to structure and illustrate the relationships of the components of degradation and restoration and their qualitative and quantitative linkages.
- *Degradation*: deterioration or impairment of the quality of a water body.
- *DPSIR*: the causal framework for describing the interactions between society and the environment adopted by the European Environment Agency (EEA): Driving forces, Pressures, States, Impacts, Responses (http://glossary.eea.europa.eu).
- *DPSIR**R***: DPSIR scheme extended by the *Recovery* components, i.e. the return of the structural and functional characteristics of the organism groups due to restoration.
- *Driver*: Social, demographic and economic developments in societies and the corresponding changes in lifestyles, overall levels of consumption and production patterns. Primary driving forces are population growth and development in the needs and activities of individuals. These primary driving forces provoke changes in the overall levels of production and consumption (EEA 2007).
- *Impact*: Impacts on human and ecosystem health, resource availability and biodiversity result from adverse environmental conditions (EEA 2007). Impact in this study refers to measurable changes of biotic conditions (of *Impact groups*) due to changing environmental states. Impact is meant neutral and can also be positive, as is the case for impact following restoration or any other *Response* measure.
- *Impact groups*: the groups of characteristics used in the WFD (Annex V, Table 1.2) to describe the ecological status using the biological quality elements: composition and abundance, diversity, sensitive/tolerant species, biomass (only phytoplankton) and age structure (only fish).
- *Mechanistic relationship*: relation between two or more objects (variables) that can be fully expressed as a formula.
- *Pressure*: Pressures include the release of substances (emissions), physical and biological agents, the use of resources and the use of land. The pressures exerted by society are transported and transformed into a variety of natural processes which manifest themselves in changes in environmental conditions (EEA 2007).
- *Recovery*: The recovery of the biota of an ecosystem or water body from the adverse impacts of environmental pressures. Recovery is expected in consequence of appropriate response measures and activities (e.g., physical *Restoration*, waste water treatment).

- *Rehabilitation*: Activity to improve the (ecological) status of degraded waters. Unlike *Restoration*, rehabilitation does only aim to partially restore or to artificially simulate the natural processes or structures in a water body. As rehabilitation does not aim to restore the natural pre-disturbance conditions, it should not be confounded with restoration (Lenders et al. 1998 in Jungwirth et al. 2002).
- *Response*: Measures taken to address *Drivers*, *Pressures*, *States* or *Impacts*. They include measures to protect and conserve ecosystem integrity and biodiversity (in situ and ex situ), and include, for example, measures to rehabilitate the impact of stressors (e.g., waste and waste water treatment) and to restore ecosystem integrity (e.g., to improve functions and processes). *Responses* may include steps taken to understand the causal chain and develop data, knowledge, technologies, models, monitoring, human resources, institutions, legislation and budgets required to achieve the target (according to EEA 2007, modified).
- *Restoration*: Activity to improve the (ecological) status of degraded waters. The goal of this process is to emulate the structure, functioning, diversity, and dynamics of the specified ecosystem. One of the most useful definitions in practice seems to be that of Henry & Amoros (1995): ‘restoration should be defined as returning an ecosystem to its conditions prior to disturbance (if known and possible), or, as in most cases, to a state as similar as possible to that which prevailed prior to disturbance, according to the changes that have occurred in the watershed’ (see also NRC 1992).
- *State*: Abiotic condition of soil, air and water, as well as the biotic condition (biodiversity) at ecosystem/habitat, species/community and genetic level (EEA 2007).
- *Statistical relationship*: relation between two or more objects (variables) that cannot be fully expressed as a formula. Instead, the relationship can be expressed as correlation or regression using a series of statistical measures to express the strength of the relationship.
- *Stressor*: used here synonymous for Pressure

Introduction

The degradation of the aquatic environment, freshwater as well as marine ecosystems, dates back for centuries and is pervasive at present in Europe (Tockner et al. 2009). Almost all river basins in Europe suffer from the impact of multiple environmental pressures: organic pollution (e.g., industrial and domestic effluents), eutrophication (e.g., due to the application of fertilisers and manure in agricultural landscapes), physical habitat and flow modification (e.g., water regulation and flood protection), and extensive water uses (e.g., cooling water, hydropower generation and irrigation). Lake ecosystems are mainly being affected by eutrophication (agricultural land use) and physical habitat modification of their shoreline, while estuaries and wetlands are mostly affected as they constitute the ultimate sink for nutrients and other sources of pollution and contaminants originating from entire river basins (Cloern 2001; Diaz and Rosenberg 2008). In addition, transitional and coastal waters are being physically modified, for instance, for flood protection purposes (e.g., Pollard and Hannan 1994) or navigation (e.g., van der Wal et al. 2002). These and other pressures might occur individually, but more often do act in combination and pose a serious threat on the ecological status of aquatic ecosystems.

The combined effects of multiple stressors render specific restoration difficult, as the stressors may or may not interact, while interaction may be synergistic or antagonistic. The combined effects of multiple stressors can be illustrated by land use effects. According to Allan (2004) **agricultural land use** degrades river ecosystems by increasing non-point inputs of pollutants, pesticides and fine sediments, by impacting riparian and stream channel habitat and altering flows. Enhanced nutrient levels and solar radiation (loss of riparian shading) lead to an increase in algal biomass, which affects the aquatic food web (e.g., increase of macroinvertebrate grazers). Major changes associated with increased urban land area include the increases of the amounts and variety of pollutants in runoff, more erratic hydrology owing to increased impervious area and runoff conveyance, increased water temperatures owing to the loss of riparian vegetation, reduction in channel and habitat structure owing to sediment inputs, bank destabilisation, scouring, channelization and restricted interactions between the river and its land margin (and floodplain).

The manifold pressures and impacts of **urbanisation** on rivers have been reviewed by Paul and Meyer (2001). The authors stress the role of ‘impervious surface cover’, which has been identified as the main *Pressure* caused by urbanisation with severe implications for the riverine hydrology and morphology (Dunne and Leopold 1978; Arnold and Gibbons 1996; Booth and Jackson 1997). McMahon and Cuffney (2000) reported the catchment’s cover of impervious area to be the major predictor of urbanisation and urban impacts on streams. Furthermore, Paul and Meyer (2001) refer to three groups of *State* variables (hydrology, geomorphology, temperature) and their *Impact* on two organism groups (fish, benthic macroinvertebrates).

The implications of urbanisation include the increase in surface runoff and peak discharge (Arnold and Gibbons 1996; Booth and Jackson 1997). As runoff is enhanced, channel dimensions enlarge, which in turn causes an increase in water temperature (Galli 1991). This hydromorphological and physical degradation affects the diversity and integrity of riverine fish communities (Klein 1979; Steedman 1988, Wang et al. 1997; Yoder et al. 1999) and of benthic macroinvertebrates (Horner et al. 1997; Yoder et al. 1999). This example of the impact of catchment urbanisation is partly illustrated in Annex 2; however, the Figure does not include ‘biotic integrity’ as a separate impact group. Biotic integrity is considered redundant here, as it is expressed as multimetric ‘Index of Biotic Integrity’ (IBI, e.g., Karr 1999) and, thus, amalgamates the individual impact groups used here in a (redundant) combined metric.

A more general review of the principle mechanisms by which land use influences stream ecosystems, has been compiled by Allan (2004). The author summarizes seven groups of pressures: ‘sedimentation’, ‘nutrient enrichment’, ‘contaminant pollution’, ‘hydrologic alteration’, ‘riparian degradation’ and ‘loss of large woody debris’. Sediment entry from adjacent crop land and sedimentation increases turbidity (Henley et al. 2000) and impairs habitat conditions for benthic algae, crevice-occupying invertebrates and gravel-spawning fish (Wood and Armitage 1997). Nutrient enrichment affects the autotroph’s production and biomass and results in a shift of algal composition. Decomposition processes lead to a decline of dissolved oxygen and sensitive taxa will be replaced by tolerant, often non-native species. In particular invertebrates and fish are affected by contaminant pollution (Woodward et al. 1997; Schulz 2004). Growth may be depressed, reproduction may fail and the endocrine systems may be disrupted. The hydrological alterations listed by Allan (2004) are similar to those reviewed by Paul and Meyer (2001) and are already mentioned above. Besides the loss of shading and the increase in water temperature due to the loss of riparian woody vegetation, Allan (2004) also mentions the increase in channel erosion and the decrease in sediment and nutrient trapping from surface runoff. Finally, the loss of large woody debris causes a loss of habitat and organic matter storage, all of which have an adverse effect on the diversity and community functions of fish and benthic macroinvertebrates (Gurnell et al. 1995, 2002; Stauffer et al. 2000).

As a consequence, the ecosystems lose biodiversity and functionality. Many sensitive species quickly disappear, while basic ecosystem functions (such as self purification, biomass production and decomposition) are believed to change significantly as soon as degradation becomes severe and exceeds a threshold. Biodiversity, ecosystem functions, and community characteristics (e.g., feedings types, habitat preferences, reproduction traits), are often known to react more or less specific along different pressure gradients and are, therefore, being frequently used as bioindicators within assessment and monitoring schemes (Huryn et al. 2002; Hering et al. 2004; Feld and Hering 2007; Feld et al. 2009; Borja at al. 2009a, b). The assessment of aquatic ecosystems, therefore, requires knowledge of the different impacts of numerous environmental stressors on the evenly numerous characteristics of aquatic communities: fish, benthic macroinvertebrates, macrophytes, angiosperms, macroalgae, phytobenthos and phytoplankton. Their relation to the components of ecosystem degradation can be based on ecological theory (e.g., Lake et al. 2007) and has often been tested and discussed, in particular in the huge body of literature on the Water Framework Directive (WFD) since 2000.

If the assessment reveals that a quality target (‘good ecological status’ with respect to the WFD) is not met for a specific lake or a river stretch, and that the target is unlikely to be met without further action, society's response to degradation is required. Degraded water bodies are being, for instance, rehabilitated to improve the physical habitat quality and to support the recovery of the biota, so that the quality targets will be met in the future. In other cases, waste water is being treated to reduce pollution of river, lake and coastal water effluents. Restoration, in its strict sense, goes one step further and aims at converting a water body back to its conditions before degradation occurred, i.e. the natural conditions without human impact (NRC 1992). Very often, restoration ecology is based on the same ecological theory, as was used before to identify and describe the relation between degradation and ecological quality (e.g., King and Hobbs 2006).

With regard to rivers, however, studies on the effects of restoration measures and various monitoring activities frequently reveal that the riverine communities do not show the anticipated and desired signs of recovery (e.g., Palmer et al. 1997; Jähnig et al. 2009). Similar results have been reported from lakes (e.g., Jeppesen et al. 2005) and transitional/coastal waters (Duarte et al. 2009). The relationships of restoration and its ecological impacts seem to (at least partly) differ from those identified for degradation. In other words: restoration is unlikely to be 'simply' the opposite of degradation (Moerke et al. 2004), which has been brought to the point by Palmer et al. (1997) as the "field of dreams": "If you build it, they will come"; this hypothesis all too often continues to be false as rehabilitation and restoration schemes turn out to be biologically ineffective.

Some restoration studies already imply that our knowledge about the time needed for a freshwater or marine ecosystem to recover from degradation is still limited (e.g., Moerke et al. 2004; Nilsson et al. 2005). One important gap addresses the endpoint of restoration and its possible deviance from the ecological status prior to degradation (reference). Another knowledge gap refers to the time scale needed for an ecosystem to recover from degradation. The salient endeavour of restoration ecology still is to identify and test the relationships between degradation and ecology and to transfer the findings to practical restoration (King and Hobbs 2006). Given the numerous studies on biologically ineffective restoration measures (see Palmer et al. 2010 for a review on river restoration effects on taxa richness and diversity) this statement could be extended by the equally salient endeavour of restoration ecology to identify and test the relationships between restoration and ecology. This might be much more straightforward as it does not include the loop via degradation and, therefore, does not run the risk of entering the field of dreams. A direct focus on restoration studies could also help identify the effects—and non-effects—of restoration, as has been summarised, for instance, by Palmer et al. (2010).

One means to identify and structure general relationships is conceptual modelling. In a broader sense, Conceptual Models constitute an ecological framework and can be used, for example, to structure the effect of the reduction and mitigation of environmental pressures on the aquatic flora and fauna. Well-documented and statistically proven, but also rather vague relationships can be identified and knowledge gaps become obvious. The linkages can be structured, for instance, based on the knowledge of rather qualitative or quantitative relationships between causes and effects, or the knowledge of empirical or mechanistic relationships. Such models are potentially helpful to structure and identify general effects of ecosystem restoration and possible recovery of aquatic communities. Finally, these hypotheses can be tested with real data and used to develop predictive models to forecast the spatial and temporal implications of restoration.

Such predictive models are considered extremely useful for river basin managers to identify and prioritise restoration measures based on existing quantifiable knowledge and the required information on the uncertainty of the predictions. At present, the decisions are—at best—based on adaptive management, a paradigm that has been advocated for many ecological restoration situations explicitly because of the lack of predictive ecological models (e.g., Clark 2002). Adaptive management should be derived from a learning experience and be based on the assessment of the outcome of restoration measures (Downs and Kondolf 2002; Woolsey et al. 2007). Thus, it requires a post-project appraisal of restoration measures in order to allow of this learning experience. Very often, however, monitoring and assessment of the progress and success of restoration measures are replaced by a rather inefficient learning experience: trial and error (Downs and Kondolf 2002).

This document aims at presenting the rationale, development and application of Conceptual Models and thereby, constitutes the basis for the development of predictive (empirical and statistical) models of the effect of river *restoration and rehabilitation* measures (in the following referred to as ‘restoration’). The report exemplifies how hypotheses on the effect of restoration can be derived from the conceptual models and how they might be tested using existing data. The specific objectives are:

- to present a general framework for conceptual modelling
- to review the existing restoration literature according to this framework
- to summarise the existing knowledge on the biological effectiveness of restoration case studies: the improvement of the i) water quality, ii) in-stream habitat structure and iii) longitudinal connectivity
- to identify well-documented and less well-documented linkages of restoration measures and biological recovery
- to identify quantitative linkages that might be subject to statistical modelling in order to predict the effects of restoration measures on the in-stream biota

The general framework for the development of Conceptual Models

The *Driver-Pressure-State-Impact-Response* (DPSIR) scheme provides a framework to link socio-economy with ecology (EEA 2007) and has been applied in previous similar studies (Elliott 2002; Karageorgis et al. 2005), whereas a main advantage of the scheme is its simplicity that renders the communication with non-scientists feasible (Stanners et al. 2007). This may be illustrated by the following narrative example.

Society's food demand, for instance, is a *Driver* of agricultural land use. The intensive application of fertilisers and pesticides in agricultural crops is often linked with pollution and eutrophication (*Pressure*) and causes water quality deterioration of adjacent rivers and lakes. Nutrients (N, P) and contaminants are being transferred with surface runoff from agricultural areas and through nutrient leaching from the soils. This inevitably has a stimulating direct effect on the growth of macrophytes and algae, but will also indirectly and negatively affect the aquatic fauna (fish, benthic invertebrates) as soon as decomposers start depleting oxygen and causing water quality deterioration (*State*). In parallel to eutrophication and contamination, rivers in agricultural landscapes are morphologically modified and hydrologically regulated (*Pressure*). As a result, microhabitats and flow regimes may change (*State*).

As a result of high population density and its demand for food (*Driver*) weirs and dams (*Pressure*) are built to control the ground water levels (*State*), but also disrupt the longitudinal connectivity of the system (*State*). Land use is often extended to the river banks and inhibits the development of a natural (vegetated) riparian buffer. As a consequence, the riverine fauna and flora is being disrupted, sensitive taxa disappear (*Impact*), and a few tolerant taxa become dominant in the system (*Impact*). Rivers and estuaries are easily being invaded by alien species (*Impact*).

To reverse degradation and to improve the ecological status, restoration and mitigation measures are required. Best-practice agriculture (*Response*), for instance, might reduce the amount of fertilisers applied per area to the amount that is equivalent to the plant biomass produced per area. Hydromorphological conditions might be actively restored (*Response*) to a more diverse habitat and flow regime. Land use in the riparian zone might be abandoned (*Response*) to promote the natural development of a diverse riparian corridor, i.e. a mixed buffer strip with grasses, shrubs and trees.

The example shows that the DPSIR scheme might be useful to structure and communicate causes and effects of degradation as well as society's response in socio-ecological systems. However, the *Response* component seems to be incomplete with respect to the objectives of this study. Similar to the degradation side of the scheme (*Pressure-State-Impact*), a similar cause-effect chain could be expected on the Management (restoration) side, i.e. a *Response-State-Impact* chain (Figure 1). A specific restoration measure or any other kind of ecosystem management is considered to have a positive effect on environmental conditions (*State*), which in turn should have a positive *Impact* on the biota, i.e. *Recovery*. In its strict sense *Recovery* refers to the full recovery of both community structure and function accompanied by all physical and chemical conditions prior to degradation (Henry and Amoros 1995). The extension of the DPSIR scheme with *Recovery* eventually results in the DPSIRR scheme, i.e. the *Driver-Pressure-State-Impact-Response-Recovery* chain.

The focus of this study is on the part of the scheme illustrated in Figure 1. In particular, we were interested in the effects of river restoration and management measures on physico-chemical, hydrological, and morphological conditions (*State*) and eventually on the *Recovery* of the in-stream flora and fauna (*Impact*).

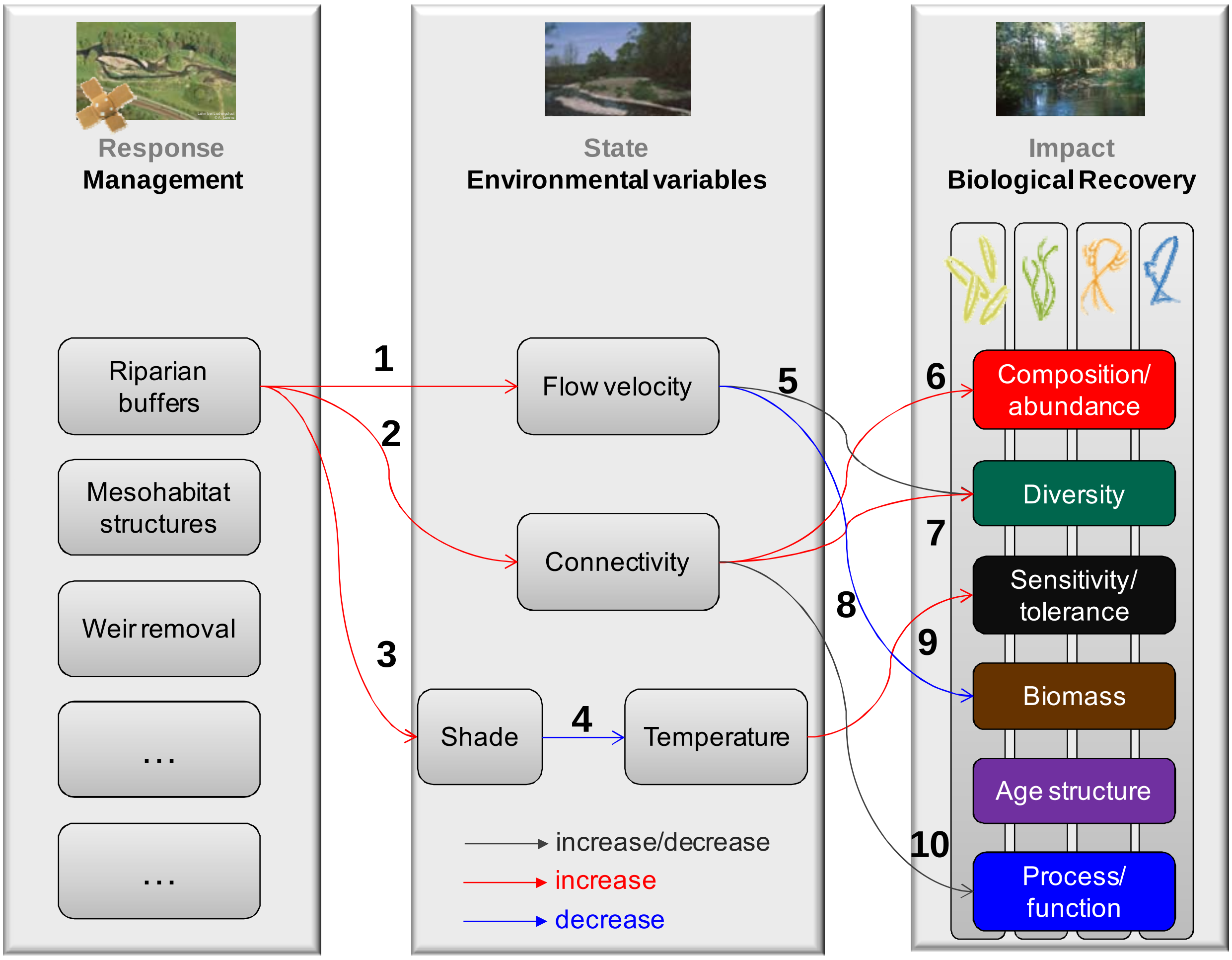

Figure 1: General Conceptual Model of Response-State-Impact chains. This chain reflects the cause-effect relationships of management and restoration. The linkage between management measures and community recovery is likely to be of indirect nature (i.e. via environmental State variables), but direct linkages may be possible, for instance, with bio-manipulation and fish stock management. Each link in the Conceptual Model refers to a relationship referenced in the reviewed restoration literature. This relationship may be either positive (red arrows), negative (blue arrows) or ambiguous (black arrow). The recovery characteristics refer to those defined by the WFD (Annex V), amended by the group of biological measures of processes and functions. See the text for further explanations.

Model components

Conceptual Models can be developed almost arbitrarily complex. A single restoration measure might mitigate the effects of several pressures in parallel and thus have various effects on water quality and physical habitat states, which ultimately control the community's change with respect to various aspects, such as community composition and abundance, functional measures, or the presence and absence of specific taxa. More complex examples might result, if multiple management measures and their ecological impacts are to be considered in parallel, which may easily end up in messy and thus largely useless illustrations. In order to limit this complexity on beforehand, we limited our study to three well-defined and common management (*Response*) measures and defined a joint structure for the recovery variables (Figure 2). A separate Conceptual Model was developed for each management measure (see Annex 1).

Management measures (*Response*)

Water quality improvement by riparian buffers in high- and low-energy streams 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).

The enhancement of in-stream mesohabitat structures aims at increasing mesohabitat diversity and is considered to promote biological diversity (Palmer et al. 2010). In particular the introduction (or omission of removal of) large wood (LWD) is considered to provide a key habitat for fish and benthic macroinvertebrates (e.g., Roni and Quinn 2001; Kail et al. 2007) that may also stimulate habitat diversity (e.g., creation of pools) by enhancing more diverse hydrological conditions (e.g., Baille et al. 2008).

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), while secondary effects can be expected on flow conditions and sediment particle size upstream and water temperature up- and downstream (e.g., Bednarek et al. 2001; Hart et al. 2002).

State variables

Environmental variables that are reported to change due to the effects of a management measure are summarised as state variables. The linkage between management and recovery might be via one single state variable (e.g., chain 1-5 in Figure 1) or via several interrelated state variables (e.g., chain 3-4-9 in Figure 1).

Structure of recovery variables (*Impact*)

Altogether six groups were applied to the four organism groups (fish, benthic invertebrates, macrophytes and phytobenthos) to structure biological *Recovery*.

- composition and abundance (e.g., number of taxa, total community abundance),
- sensitive and tolerant taxa (e.g., number of salmonid fish species, number of Ephemeroptera-Plecoptera-Trichoptera [EPT] taxa among benthic invertebrates, abundance of red-bodied chironomid taxa),
- diversity (e.g., taxon richness, diversity indices),
- age structure (e.g., relative abundance of young-of-the-year, larval development in benthic macroinvertebrates),
- biomass (e.g., fish catch biomass, phytoplankton biomass/biovolume expressed as Chl *a*) and
- processes and functions (e.g., species traits such as feeding types or body size, or measures of primary production and decomposition).

Linkage of the components

Linkage type

Each cause-effect linkage, i.e. a linkage between two objects in the Conceptual Model, is illustrated by an arrow (Figure 1). Thus, each arrow represents a relationship proven by the reviewed restoration literature. The arrows are numbered consecutively and are referred to in the literature review database (see chapter Literature review). Red arrows indicate a positive relationship, i.e. the effect variable at the tip of the arrow increases if the cause variable increases. Blue arrows indicate a negative relationship, while black arrows reflect ambiguous or unknown relationships.

The linkages were further distinguished according to their quantitative or qualitative nature.

- A quantitative linkage refers to relationships with information on the degree of change (e.g., an increase of x by 10% causes an increase of y by 20%). Linkages that are based on either empirical (e.g., x and y are correlated with an $R^2 = 0.79$) or statistical/mechanistic relationships (e.g., y is a times x) were also considered quantitative. Quantitative relationships are considered superior to qualitative and were rated stronger.

- A qualitative linkage was given, if only the direction of a trend was reported (e.g., an increase of *x* causes a decrease of *y*). Though not likely to be useful for subsequent quantitative analysis and modelling, such information is considered important for the development of hypothesis and thus included here. Qualitative relationships too can be rated strong if, for instance, multiple references supported a qualitative linkage.

Linkage strength

The strength of each linkage was estimated based on the number of references that refer to the specific relationship (either supportive or alleviative) as well as on the 'quality' of the references. The quality rating of individual references was based on the number of site/sample replicates, the study design (before-after, control/impact, or both combined), the statistical analysis applied, the significance of results, and the geographical coverage and representativeness of the study. These criteria were not defined in more detail and can be considered subjective; however, we think that the rather broad classification into strong, intermediate and weak according to Table 1 is reasonably possible for the purpose of this study.

Table 1: Criteria to evaluate the quality of individual references in the literature. Note: The matrix does not represent a classification scheme; not all criteria had to be met in a row to rate a reference weak, intermediate or strong, respectively.

Rating of criteria	No. of replicates	Study design	Statistical analysis	Significance of relationships	Geographical coverage
weak	low (< 3 replicates)	other	none (descriptive only)	none	local study, site scale
intermediate	intermediate (3–8 replicates)	before/after design (BA), control/impact design (CI)	correlation, multivariate relationships	significant	reach scale, only one (sub-) catchment
strong	high (> 8 replicates)	full BACI design	statistical modelling, regression	significant	regional study, many (sub-) catchments

Review of the restoration literature

The literature survey was conducted using major web databases such as the *ISI Web of Knowledge* and *SCOPUS*. Hence, 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. Furthermore, a strict focus was on references from the restoration literature, i.e. either publications that specifically address active restoration or reviews thereof. Therefore, the following search terms were used: *restoration* OR *rehabilitation* AND *riparian vegetation* OR *riparian buffer* OR *large wood* OR *LWD* (= large woody debris) OR *habitat structure* OR *bed structure* OR *channel structure* OR *weir removal* OR *dam removal* AND *fish* OR *invertebrates* OR *macrophytes* OR *phytobenthos* OR *algae* AND *river* OR *stream*. The search terms were used in different combinations to make sure that all relevant literature was found, but always with the limitation to the restoration/rehabilitation literature.

Publications on general ecological relationships, for instance, studies comparing natural streams (e.g., with intact riparian vegetation, control) with degraded streams (e.g., with degraded or without riparian buffers strips, impact) were only considered, if they provided strong and generalised evidence based on comprehensive data representing a broad geographical extent. All studies had to provide information on further criteria listed in Table 2 in order to be included in the review database. The criteria include those recently reported by Miller et al. (2010), who published a sound and statistical meta analysis of the literature on a similar topic.

Table 2: Attributes of restoration studies compiled in the literature database. See Annex 2 for the data sheets.

Column code	Attribute	Explanation/example
A	Serial No.	Consecutive number
B	Model code (e.g. RHhabi1 for high-gradient river and hydro-morphological restoration, model 1)	Unique code to refer to the model addressed by the reference: 1st letter: **R**iver, **L**ake, **T**ransitional/coastal; 2nd: **H**igh gradient/**L**ow gradient river; **D**eep lake/**S**hallow lake; **E**stuary/**C**oast; followed by 4 letters addressing the kind of restoration/rehabilitation: **habi** = habitat enrichment, **wlev** = water level fluctuation, **eutr** = eutrophication, **wqua** = water quality; followed by a consecutive number to provide unique codes
C	Model link No.	Number of the link in the respective conceptual model, indicated by "X" in the respective column
D	First author	Surname, initials (e.g., Mueller HP)
E	Year	Four digits
F	Full reference	Give the full reference according to the style of J Appl Ecol
G	PDF available (Y/N)	Available PDFs should be frequently uploaded to the Intranet, section > M05 > literature
H	Name of WP5.1 reviewer	Name of WISER collaborator to contact for further details/questions

Column code	Attribute	Explanation/example
I	Type of literature	Please indicate: **Re**storation, **Ge**neral ecology, **Ex**periment, **Rev**iew, **M**eta **a**nalysis, **M**o**d**elling approach (multiple choice possible, e.g., ReRv = review on restoration studies)
K	Peer-reviewed (P) or grey (G) literature	Any non-peer-reviewed literature is to be classified "grey"
L	Water body name (e.g., main river system, lake name, coastal area)	Rivers: Please indicate the main river system's name (e.g., Rhine, Danube, Mississippi), the lake name, the estuary name or the name of the coastal region
M	Size (area) of water body	E.g., lake surface area, river catchment area); the Strahler order might be given in addition for rivers
N	Region, ecoregion (e.g., Central Mountains, Western Lowlands, Alps)	Indicate the broad ecoregional/regional context, if available, but at least indicate the altitude-classification: lowland/plains, mountains, alpine
O	Continent/sub continent	E.g., North America, Europe, Australia
P	Type of restoration measure	For restoration studies: Please indicate the type of measure, e.g., dam removal, habitat enrichment, buffer strip instalment, removal of impoundments, removal of bank enforcements, bio-manipulation); measures may be categorised as: hydrology/flow, morphology/physical habitat, land use management, connectivity, maintenance
Q	Type of restoration monitoring approach	Before-after (BA), control-impact (CI), other (please specify)
R	Period between restoration and monitoring	Indicate months/years after instalment of measure
S	Brief summary of core findings	Free text to briefly outline the core contents of the reference
T	BQEs monitored	FIsh, Benthic MacroInvertebrates, MacroPhytes, Diatoms/PhytoBenthos
U	BQEs response category affected	Composition, abundance, sensitive/tolerant species, diversity, biomass, age structure, functional characteristics/processes
V	Abiotic variables measured	If abiotic variables have been recorded/measured, please specify.
W	Quantitative relationship (please specify)	Is there a quantitative relationship reported? If yes, please give the relationship (e.g., a formula)
X	Qualitative relationship (please specify)	For qualitative relationships, please specify (e.g., double increase of variable X caused decline of biological attribute Y).
Y	Estimated strength of the relationship (e.g., based on the number of datasets analysed or the number of references supporting the relationship(s))	Based, e.g., on the number of references that support the findings or the statistical power with which the findings have been proven or the amount and spatial coverage of the data analysed. **Weak** = the link is not well addressed, there is only a weak relationship reported which is not statistically proven. the study is not based on a sufficient number of replicates; **Intermediate** = the link is rarely addressed, but some qualitative (or quantitative) evidence is reported or otherwise supported by expertise, either the statistical power or the number of replicates in the study is weak; **Strong** = the link is well addressed with strong and statistically proven quantitative or qualitative evidence, the number of replicates is high which renders the result applicable to other streams or even regions.
Z	Limiting variables or attributes that may have impeded restoration effects	For instance multiple stressors not sufficiently addressed by restoration measures (e.g., insufficient water quality) or key habitats not promoted by restoration or even adverse effects of restoration (e.g., disturbance and destruction of habitats)

Of the roughly 1,000 hits reported back by the web literature databases, altogether 146 publications have been selected for a detailed review, 112 of which largely fulfilled the criteria for our review. These references are listed in (Annex 2). The final selection of references was analysed according to the attributes listed in Table 2 and converted to a MS EXCEL© database for further descriptive analysis. The studies primarily originated from North America (U.S.A.) where 62% of the studies were situated, followed by Europe (19%) and New Zealand (10%) (Figure 2). The majority of references was published after 2000 (70%), while 43 studies (38%) date back five years or less (Figure 3).

A substantial sub-set of information derived from the literature was used to develop the Conceptual Models (Table 2, "Model link No."). Therefore, each change of a *State* variable that could be attributed to an active restoration measure (*Response*) and each related change of a subsequent *State* variable or a biological impact variable was allocated a consecutive number. The number was used to number the corresponding arrow in the Conceptual Model (compare Figure 1), which then established the linkage of conceptualised cause-effect chains with evidence from the restoration literature.

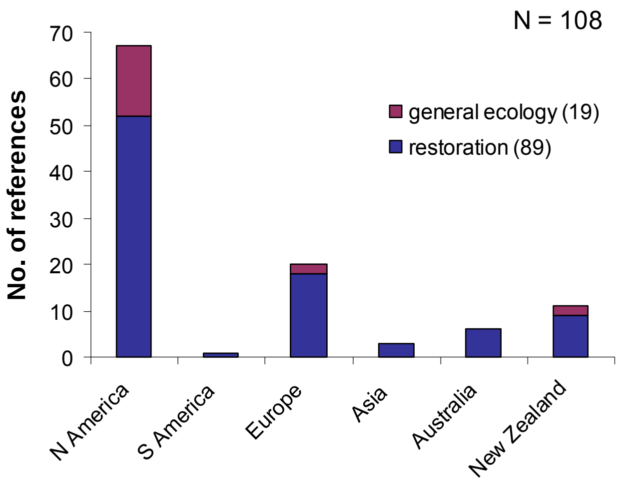

Figure 2: Origin of 108 out of a total of 112 references that provided information on the origin.

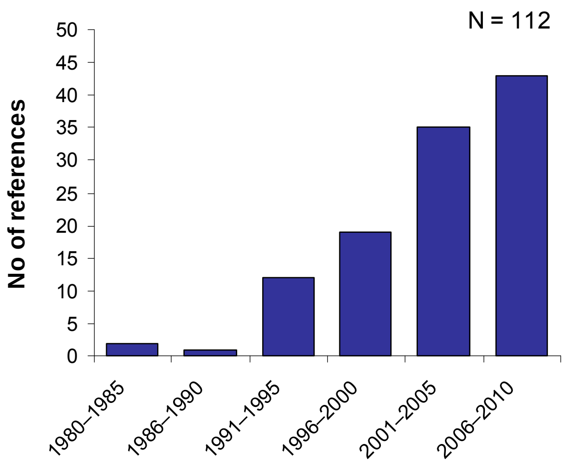

Figure 3: Time of publication of 112 references analysed.

The Conceptual Models presented in the following show all relationships of management measure, its effects on state variables and eventually the recovery of in-stream organisms. It should be stressed, however, that no reference in the literature provided statistically proven evidence of an entire cause-effect chain from the *Response* measure via one or several environmental *States* to the biological *Impact*. Most studies were limited to measuring environmental effects of a *Response* measure and only very few studies measured biological *Impacts*, yet which were not attributed to changing environmental *States*. Consequently, the Conceptual Models presented in the following reflect an overall constructed summary of the state-of-the-art derived from the literature.

Conceptual Models of river restoration

Altogether three conceptual models have been developed and referenced for this study: i) water quality improvement by riparian buffer instalment, ii) enhancement of in-stream mesohabitat structures by introduction of substrata and mesohabitat structures and iii) restoration of longitudinal connectivity by removal of weirs and dams. Each Conceptual Model consists of two components: a graphical illustration based on Figure 1 to illustrate the cause-effect chains as arrows and a corresponding MS EXCEL© sheet with literature references to the arrows and further information derived from the references.

The three Conceptual Models and corresponding literature databases can be found in Annex 1 and 2, respectively. Annex 1 contains five different illustrations for each model: the first shows the complete model with all arrows, while the arrow thickness is equivalent to the number of references that address a linkage. This illustration facilitates the identification of well-referenced cause-effect chains as reported in the literature. The four subsequent model illustrations show the linkages referring exclusively to each of the four organism groups. These illustrations may facilitate the detection of well-references organism groups and show the differences between them.

A more detailed analysis of the Conceptual Models and the literature survey is presented in the following. The main objectives were i) to summarise the existing knowledge on the biological effectiveness of management measures, ii) to identify well-documented and quantitative linkages of management measures and biological recovery and iii) to identify knowledge gaps. The following questions are addressed in particular:

- Are cause-effect chains detectable from the Conceptual Models?
- Which organism groups and group attributes showed recovery after restoration?
- Is there evidence for strong qualitative or quantitative linkages between management measures and biological recovery in the literature?
- What is the time-scale of recovery, i.e. what time is needed after a management measure to show recovery effects?
- Does the literature provide examples of failure and limiting factors that might explain why restoration had no or even adverse effects?

Water quality improvement by riparian buffers in low-energy streams

Forty-eight references met the review criteria and were transferred to the database, 35 of which represented active restoration studies and 13 additional papers on more general riparian buffer studies. Seventy percent of the studies were published after 2000. Among the restoration studies, only one reference contained quantifiable results on the relationship between annual leaf litter standing crop and annual secondary production of benthic macroinvertebrates (Entrekin et al. 2009).

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). In general, 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 are highly variable in the restoration literature. Based on their review, 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.

Results of the minimum buffer length are less frequent in the literature. 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 16 studies that also provided information on the length of the studied sites or reaches (Figure 4), this was less than 500 m for two thirds of the studies; four references had study sites >1 km length.

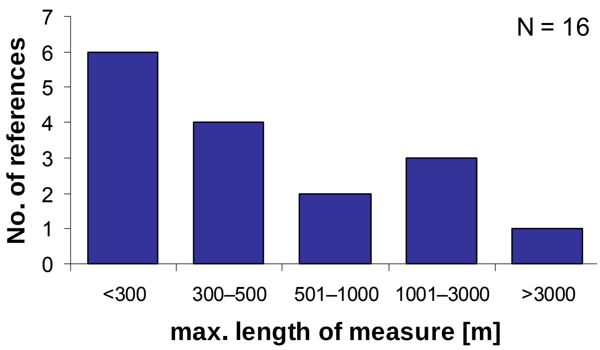

Figure 4: Lengths of study reaches provided by 16 restoration studies.

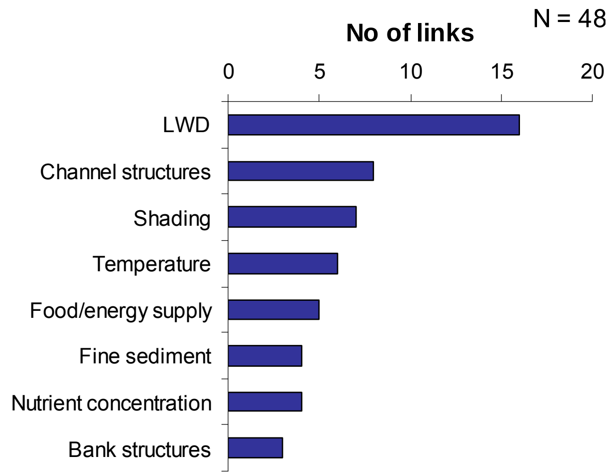

Figure 5: Ranking of most important environmental State variables based on the number of linkages (arrows in Annex 1.1) connected to them. N = number of literature references.

Are cause-effect chains detectable from the Conceptual Model?

The complete model (Annex 1.1) reveals fairly complex relationships between the restoration of riparian vegetation, its environmental effects and eventually its impact on in-stream plant and animal communities. 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 (Annex 1.1). The complexity and interrelation of these state variables is illustrated in Figure 5, based on the number of linkages (arrows) heading to and from the respective state variables.

Consequently, there is evidence from the restoration literature that riparian buffer instalment is an effective management measure to increase in-stream 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).

Which organism groups and group attributes showed recovery after restoration?

As already stated before, there is comparatively little evidence in the literature 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 underlying 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. The cited studies refer to the link numbers 3, 32, 33 and 34 in Annex 1.1.

In contrast, the Conceptual Model revealed a considerable number of relationships between environmental *States* and biological *Impact* (Annex 1.3). Altogether, 14 links to benthic macroinvertebrates can be derived from 34 references in the reviewed literature (Figure 6). Most studies reported changes of benthic invertebrate community composition and richness (70%), while only four studies addressed functional aspects of the community. Interestingly, six studies reported effects on benthic invertebrate biomass and age structure (larval development; e.g., Lester and Boulton 2008; Entrekin et al. 2009), two aspects which are not covered by WFD-compliant monitoring. The Conceptual Model revealed four predominant *State* variables impacting benthic macroinvertebrates: fine sediment, water temperature, food/energy supply (particulate organic matter) and large wood.

Seventeen 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).

Considerably less often were aquatic macrophytes and phytobenthos addressed in the literature (compare Annex 1.4 and 1.5, and Figure 6)—although the direct relationships between shading, sediment particle size and nutrient enrichment, and aquatic plants are obvious.

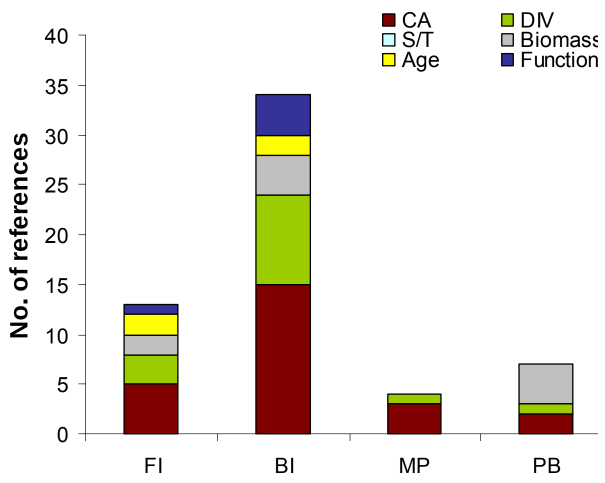

Figure 6: Number of references addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB). As a study may refer to more than one community attribute, the overall number of references exceeds the number of 35 reviewed restoration references.

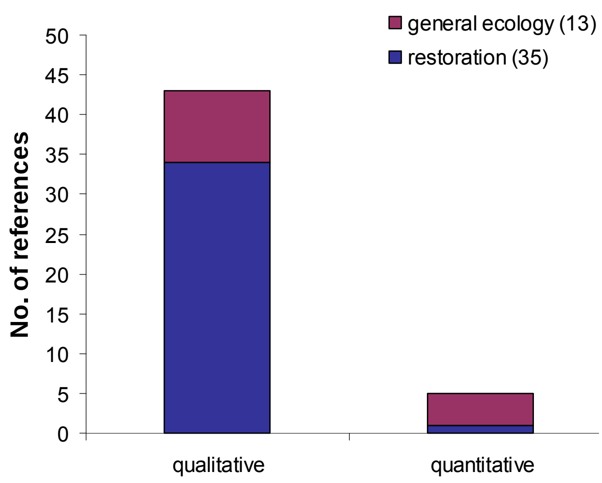

Figure 7: Relation of qualitative to quantitative linkages reported in the restoration and general ecological literature. (Numbers in brackets indicate total number of references reviewed).

Is there evidence for strong qualitative or quantitative linkages?

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 3). 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 show 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 3: 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) W = buffer width k = constant (50 ft) 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; $p < 0.001$, $r^2 = 0.80$; F (No./100 m) = (0.1326 * stand age) + 7.3952; $p < 0.001$, $r^2 = 0.63$

Nevertheless, there is a clear lack of reference for strong relationships to the in-stream 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 reviewed for this study report qualitative results. This is useful 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) published 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.

To summarise the findings, 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.

What is the time-scale of recovery?

A satisfactory answer to this question is hardly possible based on the review presented here. But some theoretical considerations may show how long riparian buffer management can take to achieve maturity and to provide all relevant ecological functions. Native 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 the full set of functions with respect to nutrient and sediment retention, and temperature control. However, this time frame is also likely to be 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).

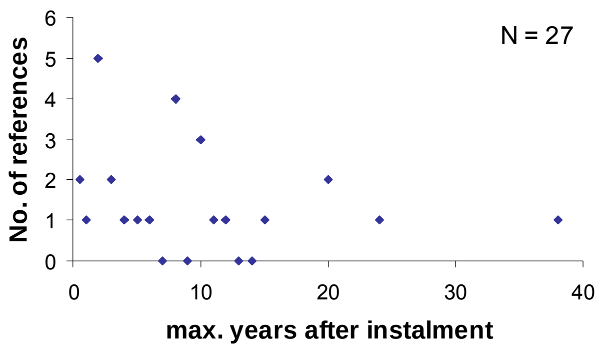

Figure 8: Timing of 27 reviewed studies relative to the time of instalment of restoration measures.

Twenty-seven references provided information on the timing of field surveys relative to the instalment of restoration measures (Figure 8). The majority of studies was conducted less than 15 years after instalment, which might be sufficient to detect the effects of in-stream habitat improvements, but which is likely to be insufficient to detect major 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.

Does the literature provide examples of failure and limiting factors?

Only three out of 35 restoration studies analysed 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 found that in-stream nutrient concentrations have not decreased although the mean forested buffer density in the 15 stream reaches increased from 33 to 44%. The authors attributed this failure to insufficient buffer age, width and connectivity (gaps in the buffer largely reduced its 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 effect measurable for the benthic invertebrate community. The authors assumed that the re-vegetated buffers require much more time to show anticipated positive effects on the biota.

Another study showed strong effects, but implied the limited transferability of results to other regions. Warren et al. (2009) quantified large wood loading to 28 streams in the northeastern United States with a 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 3). The application of their models to other regions, however, revealed that the regression models cannot be applied to data from other regions because of regionally different forest characteristics and the legacy of forest land use.

Enhancement of in-stream mesohabitat structures

Among the plethora of published scientific papers on remediation effects the studies of meso-habitat enhancements at mountainous "high-energy" rivers were well represented (see references cited in Roni et al. 2008; Miller et al. 2010; Palmer et al. 2010). The type of measures ranged from improvements of single in-channel structures (e.g., Brooks et al. 2002) to rebraiding of river sections (e.g., Jähnig et al. 2009). However, the scarcity of references on remediation success was striking. For the high-energy rivers we found only twelve papers that at least revealed significant qualitative responses of the aquatic communities to restoration efforts. These publications are evaluated in the following. Most representative papers informing about restoration failures are reviewed in the latter section of this chapter. The corresponding Conceptual Model is shown in Annex 1.6.

Nine publications documented in-stream treatments in the U.S. that feature a comparatively dense network of restoration monitoring (Bernhardt et al. 2005). Especially habitat enhancements in the states of California, Washington and Oregon were well represented. Only little biological response to meso-habitat restorations was reported from high-energy rivers in Europe. The reviewed papers covered an array of stream types ranging from small-sized, steep gradient brooks at alpine elevations to 6th order mountain rivers with channel widths exceeding 50 metres. While most studies examined the effects at only one or two individual locations, three papers reviewed the influences of restoration for 30 different water bodies within larger geographical areas (Roni and Quinn 2001; Binns 2004; Roni et al. 2006).

Half of the studies employed "Before-After-Control-Impact" designs (Smith et al. 1993). These publications implemented only one or two control locations despite the statistical vigour of asymmetrical sampling designs with multiple replicates (Underwood 1994). The analyses of other papers built on "Control-Impact" (or post-treatment) designs, often arguing that sampling prior to restoration was unfeasible. The lengths of the treated reaches showed a considerably broad range, spanning from 75 metres at small, steep-gradient brooks in the western U.S. to two kilometres at the 6th order Alpine Drau river in Austria. The average length of restored sections derived from eleven literature sources amounts to approximately 100 metres (median value). The class frequencies of publications specifying the section lengths are summarized in Table 4.

Table 4: Class frequencies of restored section lengths as given by eleven reviewed papers

Length of restored section [m]	Number of reported restoration sections
≤150	36
151-999	8
≥1000	4

One of the most effective measures was the placement of large woody debris (LWD) that was evaluated in half of the reviewed papers (Riley and Fausch 1995; Cederholm et al. 1997; Hilderbrand et al. 1997; Gerhard and Reich 2000; Roni and Quinn 2001; Moerke et al. 2004). Other improvements of the river structure comprised the instalment of boulders (Roni et al. 2006), the removal of bed and bank fixations and the widening of the channel (Jungwirth et al. 1995; Muhar et al. 2008). Several publications investigated the aquatic communities in response to a combination of measures, namely the application of the "natural channel design" scheme according to Rosgen (1994) and Baldigo et al. (2008) or the realignment of entire stream courses (Herbst and Kane 2009; Moerke et al. 2004).

The majority of studies dealt with the impact of stream habitat enhancement on fish (especially salmonids). Effects on benthic invertebrates or phytobenthos were rare. Studies on LWD treatments generally revealed enhanced abundances of key fish species (especially salmonids) that benefited from pool enlargements and higher pool frequencies. Significant effects on species richness or diversity due to log placement were less obvious from the literature.

Are cause-effect chains detectable from the Conceptual Model?

Even if based on a limited number of references reviewed, there are some cause-effect chains detectable from the Conceptual Model. Annex 1.6 depicts various pathways specifying the impacts of restoration measures on the abiotic stream environment and the aquatic communities. Especially the effects of large wood (LWD) placement are well documented (e.g., Riley and Fausch 1995; Hilderbrand et al. 1997); fallen trees in the channel decrease the stream's riffle-pool-ratio and, thus, enhance the frequency and size of suitable deep water habitats for salmonids. These habitat changes cause an increase of trout and salmon biomass that may be only of seasonal significance (Cederholm et al. 1997). Furthermore, the high pool abundance fosters juvenile fish and affects the community's age structure.

Substrate diversity is directly enhanced by the addition of LWD, but also by the placement of mineral substrates such as boulders (Jungwirth et al. 1995; Gerhard and Reich 2000). Channel realignment can affect substrate diversity, if coarse mineral material is being added (Moerke et al. 2004; Herbst and Kane 2009). The effects resulting from these measures are manifold; diversity and abundance of benthic invertebrates increase (Gerhard and Reich 2000; Moerke et al. 2004), more Ephemeroptera-Plecoptera-Trichoptera (EPT) taxa occur and the ratio of shredders is enhanced (Herbst and Kane 2009). The fish community reveals augmented taxonomic diversity (Jungwirth et al. 1995).

Which organism groups and group attributes showed recovery after restoration?

Several studies consistently concluded that an increased pool-riffle ratio was favourable to the densities of (salmonid) fish (see also Annex 1.9) at the restored sections (e.g., Riley and Fausch 1995; Cederholm et al. 1997; Roni et al. 2006). Roni and Quinn (2001) provided quantitative relationships between LWD density, pool area and the abundance of Coho salmon (*Oncorhynchus kisutch*) during summer. A ten-fold increase of the LWD density, for instance, resulted 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 the opposite 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 is 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 river bed and the initiation of type-specific in-stream structures at the local and reach-scale. The rehabilitation initiated an improvement of the ecological status of the fish community up to one 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 four papers on invertebrate responses (see also Annex 1.8) 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 low 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.

The two publications studying the effects of complete river realignments documented short-term responses of invertebrate richness and abundance. Two years after treatment the number of EPT (Ephemeroptera-Plecoptera-Trichoptera) taxa increased by factor seven, leading to an overall increase of EPT/total taxon richness (Herbst and Kane 2009). Tolerant organism abundance decreased by more than 15%. The abundance of sensitive organisms was enhanced by almost 10%. The share of feeding types changed towards higher ratios of shredders. Moerke et al. (2004) showed that within one year after restoration, major trophic groups (benthic algae, benthic macroinvertebrates and fishes) recovered to or exceeded levels in the degraded, unrestored reach. Five years after the restoration macroinvertebrate density remained higher in the restored reaches, whereas macroinvertebrate diversity in the restored reaches were similar to or below levels in the unrestored, channelized reach.

Although phytobenthos (see Annex 1.7) communities are generally not expected to respond to habitat enhancement measures, 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 resulting from log placements caused the elevated biomass values.

Is there evidence for strong qualitative or quantitative linkages?

The linkages between measures and response on the level of habitat or biological community were mostly described in qualitative terms. The treatments generally resulted in enhanced aquatic habitat heterogeneity. The publications reported increases of variability of channel width and depth, bed substrate diversity, flow velocity and augmented riparian shelter. Positioning of LWD, for instance, affected the meso-habitat structures of the treatment reaches with regard to the number and size of pools and riffles. Hilderbrand et al. (1997) demonstrated that 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. These findings, however, were constraint to the "low-gradient" stream (gradient of < 1%) investigated by the authors. The "high-gradient" study site did not reveal significant differences. Gerhard and Reich (2000) showed high correlations of the amount of LWD and the proportion of semi-aquatic areas (e.g., sand and gravel bars, maximum r = 0.8).

Significant response patterns were reported for various biological parameters (Table 5). Some studies inferred quantitative relationships, namely for hydromorphological parameters and various attributes of the fish community. These examples—described in the previous section (Jungwirth et al. 1995; Roni and Quinn 2001; Roni et al. 2006; Muhar et al. 2008)—present moderate to strong correlations between mirco- and mesohabitat, in-stream features and the abundance, richness and diversity of fish, which provides evidence for the effectiveness of in-stream habitat enhancement to improve the fish community. Nevertheless, we advise caution with the usage of such relationships for predictive modelling, as the results are often based a small numbers of replicates and cannot be transferred to other regions without further testing.

Table 5: 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 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.	NFS = 0.00927 * VMD + 6.12, r = 0.86; n = 15; NFS: Number of Fish Species, VMD: Variance of Maximum Depths FSD = 0.0007014 * VMD + 1.28; 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.	

Table 5, continued.

Publication	Qualitative	Quantitative
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 SDR = -0.83 * PAR + 0.15; R^2 = 0.45, n = 20; SDR: age 1+ steelhead trout density response 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 $p < 0.05$ (**) % pool area/Coho abundance: 0.51*; % pool area/trout abundance (> 100 mm length): 0.54*; LWD/dace: -0.77**; LWD/trout abundance (< 100 mm length): -0.70**

What is the time-scale of recovery?

The time span between restoration and monitoring of effects was highly variable and ranged from one to 20 years with an average (median) value of three years (Table 6). On average, the sampling was performed only two times after restoration and then compared against the controls (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 within-year (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 sampling after restoration seem 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 realignment, but noted some decline in habitat quality over the following five years of monitoring due to insufficient sediment trapping upstream of the treatment. 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 that can lead to long-term recovery of deficient processes.

Table 6: Timing of monitoring after the instalment of in-stream habitat structures.

Monitoring after installation of measure (years)	Number of publications
1	1
2	2
3	4
> 3 (up to 20 years in one reference)	3

Does the literature provide examples of failure and limiting factors?

To demonstrate existing linkages between in-stream restoration efforts and biological responses we selected publications that documented effective treatments, however, such “success stories” about in-stream habitat enhancement are rather scarce. Many references reported activities that lead to measurable structural improvements in the short-term, but that failed to show impact on the in-stream biota. For instance, the majority of studies reviewed by Palmer et al. (2010) did not show a measurable effect of local (site-scale) measures of habitat improvement (e.g., the placement of LWD and deflectors) on benthic macroinvertebrate communities in the U.S. and Europe.

According to Miller et al. (2010), macroinvertebrate responses to restoration were documented in a “myriad of weakly replicated, inconclusive, and even conflicting published studies”. This seems to highlight some general flaws in restoration science and questions the methods to evaluate treatment effects (see also Shields 2003). Brooks et al. (2002) even argue that high within-study variability and low statistical power may render the use of 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.

Palmer et al. (2010) concluded that the overall level of watershed deterioration is relevant to the success of mesohabitat enhancement. Therefore, only whole-watershed perspectives can provide insight into whether a specific project will succeed in a specific place. They 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.

Removal of weirs and dams

Altogether, 31 references were analysed to develop the Conceptual Model on weir and dam removal conceptual (< 5 m height). Among them, fifteen papers represent active weir removal case studies, another ten review the effects of weir removal, and six additional references provide basic ecological relationships between related habitat modifications and aquatic organisms in streams. Most studies were conducted in and 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 studies compared conditions before and after weir removal (BA), while seven out of 15 restoration case studies included comparisons of control and impacted sites (BACI). Most of the studies have sampled several sites of hundred meters stretch covering total sections of kilometres.

The removal of dams and its possible ecological impacts 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 of a natural flow, i.e. 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. More recently, 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 does include 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 onto floodplain habitats (Acreman and Dunbar 2004).

Occasional floods reconnect the aquatic and riparian habitat (Shuman 1995; see also Jähnig et al. 2009 for a more recent study), 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 now possible for migrating species 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 removals may be best considered as ecological disturbances. Removal of small dams generally results in the transformation from lentic to lotic river systems upstream 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 and thus expected results should not be the same (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 illustrating the weir removal conceptual model provides little information concerning the effectiveness of the restoration. Indeed, the effectiveness is rarely measured and elements of success are very vague. However, in most of the case, 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.

Are cause-effect chains detectable from the Conceptual Model?

Weir removal is different from the cases presented and discussed before (Annex 1.10), since it is necessary to separate upstream from downstream effects to understand involved processes. Indeed, effects of weir removal are quasi inverted upstream and downstream the barrier. Also hydro-morphological processes are far better referenced than impacts of habitat condition changes on different organism groups. Looking at the number of references (arrows in Annex 1.10) supporting each linkage, four main cause-chains stand out (see Figure 9). They are all related to the release of impounded sediments to the downstream zone.

- i) a diminution of depth upstream of the removed weir while impounded sediments are released (e.g., Bushaw-Newton et al. 2002; Chaplin 2003) leads to
- ii) an augmentation of the turbidity of the water downstream (e.g., Chaplin 2003 and Hart 2003),
- iii) a diminution of sediment size downstream (e.g., Cheng et al. 2007) and
- iv) a diminution of the depth downstream as the pools are filled (e.g., Rathburn et al. 2003; Burroughs et al. 2009). These first chains are really well documented essentially because they have negative impact on BQEs communities and represent the main negative shot-term effects of weir removal (e.g., Bednarek et al. 2001; Thomson et al. 2005).

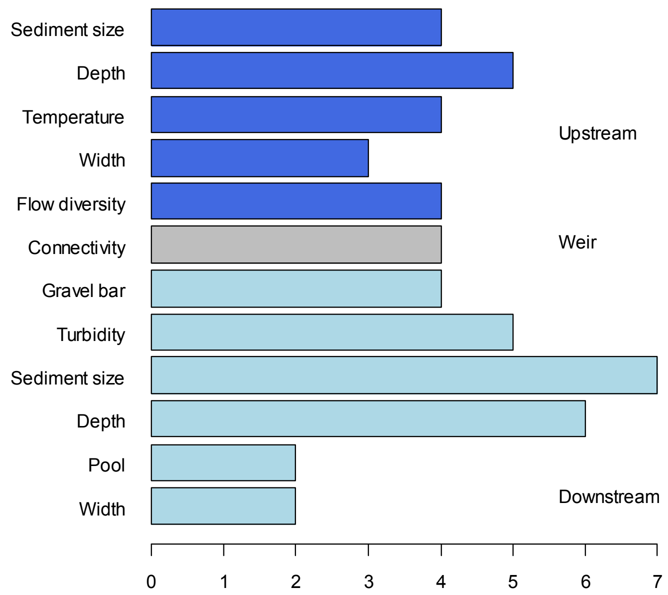

Figure 9: Most important environmental State variables based on the number of linkages (arrows in Annex 1.10) connected to them and derived from N = 17 literature references.

Four other cause-chains are less well documented, but nevertheless constitute either strong effects of weir removal.

- i) increase of sediment size,
- ii) increase of flow diversity,
- iii) decrease of water temperature (e.g., Kanehl et al. 1997; Hill et al. 1993) in the upstream zone, and
- iv) restoration of hydro-ecological connectivity (e.g., Poff 1997; Gregory et al. 2002), all of which greatly contribute to the recovery of in-stream plant and animal communities (e.g., Bushaw-Newton et al. 2002; Maloney et al. 2008). For instance, the recovery of migratory fishes on river following the reestablishment of the hydrological connectivity is a key argument for restoration and has been well documented during this last decade (e.g., de Leaniz et al. 2008).

Which organism groups and group attributes showed recovery after restoration?

More generally, the biological *Impact* of weir removal had been more studied for fish and benthic invertebrates than for macrophytes and phytobenthos (Figure 10). For the four organism groups subject to this study, effects are usually measured as abundance or species richness, but some papers consider also effects on functional measures such as benthic invertebrate feeding habits (e.g., Maloney et al. 2008) and 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 turbidity and oxygen enrichment (e.g., Orr et al. 2006; Bushaw-Newton et al. 2002).

Twenty five references in the reviewed literature support the six links related to fish *Impact* (see Annex 1.14) bringing out 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).

Twenty eight papers support the seven links related to benthic invertebrates and nine of these corroborate the negative impacts of clogging of coarse gravel river beds by fine sediment on the abundance of this organism group (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 river width and restoration of the connectivity (e.g., Shafroth et al. 2002) while modifications in phytobenthos communities abundance and composition are largely correlated to sediment size and water turbidity (e.g., Baattrup-Pedersen et al. 1999; Orr et al. 2006).

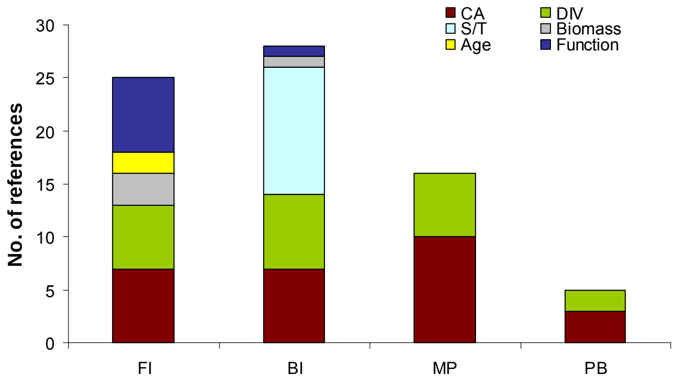

Figure 10: Number of references addressing the community attributes composition/abundance (C/A), sensitivity/tolerance (S/T), age structure (Age), diversity (Div), biomass and function of fish (FI), benthic macroinvertebrates (BI), macrophytes (MP) and phytobenthos (PB). As a study may refer to more than one community attribute, the overall number of references exceeds the number of 25 reviewed restoration references.

Is there evidence for strong qualitative or quantitative linkages?

All studies reviewed here provide qualitative analyses of the process; no reference gave quantifiable results in the sense of regression formulae or even mechanistic relationships (Table 7). Statistic or quantitative analyses based on regression and mechanistic modelling are absent from the literature, but multivariate analysis (ANOVA, PCA) is frequently being used to detect and identify patterns of the biological *Impact* of weir removal (e.g., Bushaw-Newton et al. 2002; Pollard et al. 2004; Thomson et al. 2005). However, Cheng et al. (2007) have studied the removal of the St. Johns Dam (2.2 m high) on the Sandusky river in Ohio and shown that bed deposition and scouring in the reservoir accounted for a decrease in the bed slope of 30% and bed material sizes downstream at least 40% finer than pre-removal conditions. Most of the references of the linkages are consistent and provide the same information. For instance, studies on Chipolata river, Florida (Hill et al. 1993), Manatawny creek, Pennsylvania (Bushaw-Newton et al. 2002) and Baraboo river, Wisconsin (Stanley et al. 2002) have all revealed a decrease in temperature upstream leading to an increase of dissolved oxygen. Both strong qualitative and quantitative linkages are related to sediment discharge after weir removal.

This summary of the literature shows that during the last decade substantial efforts were made to investigate the effects of and processes initiated and restored by weir removal. Nevertheless there is still a lack of quantitative measures to model and predict processes in order to estimate restoration impacts on the biota.

Table 7: 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.	

Table 7, continued.

Reference	Type	Qualitative	Quantitative
Thomson et al. (2005)	Active restoration	Downstream sedimentation following dam removal can reduce densities of macroinvertebrates and benthic algae and may reduce benthic diversity, but for small dams such impacts may be relatively minor and will usually be temporary; benthic invertebrate density was significantly lower at downstream sites after complete removal than during pre-removal or partial removal stages, but remained relatively constant at upstream sites (ANOVA); benthic invertebrate assemblages were studied using NMSD ordination method	
Cheng et al. (2007)	Active restoration	After weir removal, net sediment deposition occurred downstream of the dam, and net erosion occurred in the reservoir resulting in bed deposition and scouring in the reservoir accounted for a decrease in the bed slope of 30% and bed material sizes downstream at least 40% finer than pre-removal conditions; bed deposition and scouring in the reservoir accounted for a decrease in the bed slope of 30% and bed material sizes downstream at least 40% finer than pre-removal conditions.	
Maloney et al. (2008)	Active restoration	Following the breach, relative abundance of Ephemeroptera, Plecoptera and Trichoptera (largely due to hydropsychid caddisflies) increased upstream probably because the increased flow and particle size in former impoundments favour filter feeding taxa that cling to substrate (e.g.hydropsychidcaddiflies).	
Burroughs et al. (2009)	Active restoration	Sediment fill incision resulted in a narrower and deeper channel upstream,with higher mean water velocity and somewhat coarser substrates. Downstream deposition resulted in a wider and shallower channel, with little change in substrate size composition. Water velocity also increased downstream because of the increased slope that developed.	
Tzsydel et al. (2009)	Active restoration	Riparian and land plants developed intensively at the bottom of the Drzewieckie Reservoir immediately after it was emptied. Short-term flow fluctuations usually diminish the quality and quantity of benthos.	

What is the time-scale of recovery?

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. While some of them take years to centuries to recover, others recover in days to months. The reestablishment of the connectivity that allowed migratory fish movement is quasi-immediate whereas the full recovery of habitat could take decades. While hydromorphological parameters and water quality might recuperate in few years, biotic *Impacts* generally require several years to decades after removal, and are expected to dissipate once sediments are transported farther downstream (e.g., Thomson et al. 2005). For instance, fine sediment recovery time depends on quantity of sediment accumulated in the reservoir, on water velocity, on the gradient of the river bed and eventually on the specific technique of removal (Bednarek 2001) and may take up to 80 years.

Among in-stream organism groups the aquatic vegetation and mussels are among the slowest to recuperate as reported in the literature.

Does the literature provide examples of failure and limiting factors?

Many organisms are limited in their recovery by restricted habitat availability, which is considered to be the most important limitation factor. A recovery of habitat variability required geomorphologic processes similar to pre-dam condition (Doyle et al. 2005). For fish two cases can be considered. First, if fish communities are impacted by the physical barrier (limitation of migration) weir removal will instantaneously restore this *Impact*. Second, contrastingly, if fish are limited by the absence of suitable habitats to complete their life cycle, ecological recovery required the re-establishment of pre-removal geomorphologic and hydrologic conditions. If the geomorphologic changes are irreversible, ecological recovery of the stream reach is hardly possible.

Another limiting aspect refers to the size of the weirs/dams. Orr et al. (2006) concluded that the Impact of dam removal in Boulder Creek, U.S.A. was rather small compared with the natural variability of the entire system. This finding suggests that small weir and dam removal may not have long-term deleterious effects (see also Thomson et al. 2005).

Discussion and outlook

Evaluation of Conceptual Models

Overall, the Conceptual Models turned out to be a powerful tool to review the existing literature in a structured way. The criteria defined for the review together with the datasheets compiled to illustrate the Conceptual Models provide a sound basis for a quantitative review according to scientific standards and recent advances in meta analysis (compare Miller et al. 2010; although a meta analysis was not aimed with this study). On the other hand, the reference to the DPSIR scheme (in its more recent version published by EEA 2007), here with a strong focus on the known linkages between *Response*, *State* and *Impact*, facilitates the communication of results to practitioners. Even moderately complex models can be used to illustrate well-known cause-effect chains as well as knowledge (publication) gaps. Recommendations can be made based on the descriptive analysis of objects (boxes) and relationships (arrows) in the Conceptual Models. For example, the Conceptual Model on riparian buffers (Annex 1.1) reveals strong evidence for riparian wooded vegetation to retain nutrients and fine sediment, to provide shade and decrease the in-stream water temperature, and to structure the in-stream habitat condition by the provision of large wood.

These effects on the *State* variables have been frequently proven to have positive effects on the richness, diversity and abundance of in-stream fish and benthic invertebrates. The effects on aquatic macrophytes and benthic algae (*Impact*) turned out to be less well-studied and reported. Nevertheless, a clear focus on measures of richness and abundance is evident for all organism groups. The focus on richness and abundance might reveal the general suitability of taxa counts and densities to indicate the effects of riparian buffer restoration. But presumable, this finding is also (rather?) owed to the lack of suited indicators in restoration ecology that too account for process-related and functional aspects of the communities and eventually for the overall instream-riparian-floodplain ecosystem integrity.

The discussion on the pros of Conceptual Modelling as presented in this study should not conceal its limitations. Annex 1.1, for instance, implies numerous cause-effect chains that direct from the *Response* measure (buffer restoration) to biological attributes (Impact), primarily over one or several *State* variables. Such ‘complete’ chains, however, are rarely supported by single restoration references. Most studies reflect only a short part (e.g., one or two linkages) of such chains and, hence, the overall Conceptual Model can be considered a puzzle in which each study refers to one or a few pieces. Nonetheless, we believe that the overall models provide sufficient evidence for the effectiveness of such constructed cause-effect chains to help guide future restoration efforts. Moreover, the models do not present a final state; future monitoring of, and scientific studies on restoration will help adapt and improve the Models.

The general applicability of our Conceptual Model facilitates its application also in other aquatic ecosystems covered by the European Water Framework Directive. This has been extensively practised during an expert workshop[1] attended by 20 river, lake and marine scientists. Besides the results presented here, two examples on i) the management of lake water level fluctuation and ii) the rehabilitation of lake eutrophication have been constructed and discussed. These results are subject to a joint manuscript together with the findings presented in this study. The applicability to marine systems (transitional and coastal waters) has been discussed, but Conceptual Models on cause-effect chains for marine ecosystems have not been constructed yet.

Further fields of application and extension of Conceptual Models

It is obvious that Conceptual Modelling potentially provided more insight in the structure and evidence of ecosystem degradation, too. Degradation ecology is much more advanced than restoration ecology, which is evident from the huge body of literature available on the effects of environmental stressors on ecosystem fauna and flora. Degradation ecology has built the basis for numerous attempts to assess the ecological integrity of ecosystems using biological indicators (e.g., Zelinka and Marvan 1961; Karr 1999; Karr and Chu 1999; Hering et al 2004; Furse et al 2006). These attempts date back more than 100 years, while restoration ecology can be considered a comparatively novel branch in applied ecology that came up in the 1990s (Figure 3). In fact, the previous version of this study also aimed at developing Conceptual Models of degradation (see draft WISER Deliverable 5.1-1 available at www.WISER.eu). This plan, however, was abandoned since degradation models do not directly provide evidence on real and measurable effects of restoration. They may help water managers understand the causes and effects of degradation and develop potential measures to mitigate those effects, but they rarely provide evidence to facilitate effective restoration. Restoration is not the opposite of degradation (Moerke et al. 2004).

The simple structure of our Conceptual Models facilitates a broad field of application in ecology, also beyond the scope of this study. Other *Responses* (e.g., grassland management, reduction of emission of industrial pollutants) could be easily integrated, as well as other organism groups (*Impact*, e.g., birds, humans) and their attributes (dispersal, growth). Moreover, the linkage to the DPSIR scheme renders Conceptual Modelling also suitable for application in socio-economic sciences, for instance, to study the direct and indirect effects of market regulation and corresponding economic policies (*Response*) on consumer behaviour (*Impact*), for instance, changing consumption of fossil fuel and other products harmful to the environment.

[1] The workshop on Conceptual Modelling of restoration was held at ALETRRA, Wageningen, in November 2009. See www.WISER.eu for a brief summary. The results of the workshop provided the basis for this Deliverable 5.1-1.

A further classification of *State* variables could help derive aggregated information form Conceptual Modelling that is not referred to in our study. A further classification of environmental variables into spatial (site, reach, catchment) and temporal (short-term *vs.* long-term) scales could facilitate the identification of spatial hierarchies of *States* and, in turn, could help define appropriate sequences or combinations of measures to account for natural processes and hence maximise the effectiveness. Large wood, retention and temperature, for example, are key variables in the riparian buffer model (Annex 1.1). Temperature and retention effects of riparian vegetation, however, are likely to be scale-dependent (e.g., Sutton et al. 2009) and unlikely to have measurable effects at the site scale (several tens to one hundred metres). Rather vegetated buffer must span over several hundreds of metres or even several kilometres to effectively reduce eutrophication, sedimentation and warming. Contrastingly, local habitat structure and diversity would be significantly increased by natural recruitment of large wood from such buffers. Accordingly, reach-scale water quality improvement requires a scale of restoration different from that required for local habitat effects. In this example, only reach-scale buffer instalment (or catchment-scale buffer management) would meet both requirements as in-stream and riparian processes basically follow a top-down controlled hierarchy, i.e. from the catchment to the site scale (see Beechie et al. 2010 for a recent summary of process-oriented river restoration).

A further sub-division of States into relevant physical, chemical and biological processes would help identify linkages of *Response* and processes and of processes and *Impact*. Hence, Conceptual Modelling might be extended and modified to account for more recent advances in restoration ecology aiming at the re-establishment and restoration of key ecosystem processes and functions (e.g., Beechie et al. 2010), such as natural flow, erosion and deposition, supply of organic matter to the in-stream food web, primary production and decomposition and self purification. Process-based principles of river restoration could help design restoration schemes to better account for the scale of the environmental problem and to meet the physical and chemical potential of a site or reach.

The replacement or extension of Impact groups to measures of biodiversity and ecosystem services would open the field of application of Conceptual Modelling to current aspects of biodiversity policies. Ecosystem service provision (e.g., provision of food, clean water and air, nutrient cycling) is considered to be closely linked to biodiversity, and both are largely threatened by ongoing ecosystem degradation at the global scale. Approximately 60% (15 out of 24) of the ecosystem services examined during the Millennium Ecosystem Assessment (MA 2005) are being degraded or used unsustainably, incl. fresh water and water purification. While the knowledge about the causes of degradation and ecosystem service loss is comparatively advanced, there are two major drawbacks that render the protection and restoration of ecosystem services and of underlying biodiversity (rates) difficult. First, there is a clear lack of suitable indicators capable of detecting the aspects of biodiversity that control ecosystem services (Feld et al. 2009; 2010). And second, suitable and wisely applicable indicators of ecological ecosystem services (e.g., self purification, nutrient cycling) are still to be developed.

Consequently, there is still an incomplete and insufficient knowledge about the impact of ecosystem degradation, while ecosystem restoration to protect and restore ecosystem services at natural rates is still in its infancy. The Conceptual Modelling presented in this study might help identify and structure the existing knowledge about this important aspect of ecosystem restoration. If combined with scaling and process-based aspects, such models might provide the basis for integrated ecosystem restoration and management to address the various corresponding policies in parallel. This synergy could foster integrated river restoration and management in line with the demands of the European Water Framework Directive, the European Habitats Directive, the Convention on Biological Diversity and related national and regional policies.

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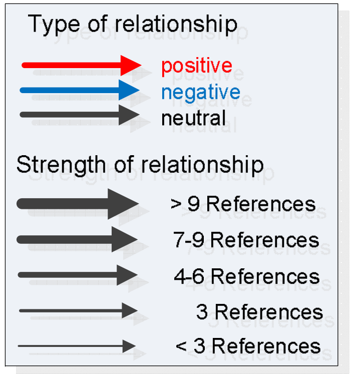
Annex 1: Illustrations of Conceptual Models

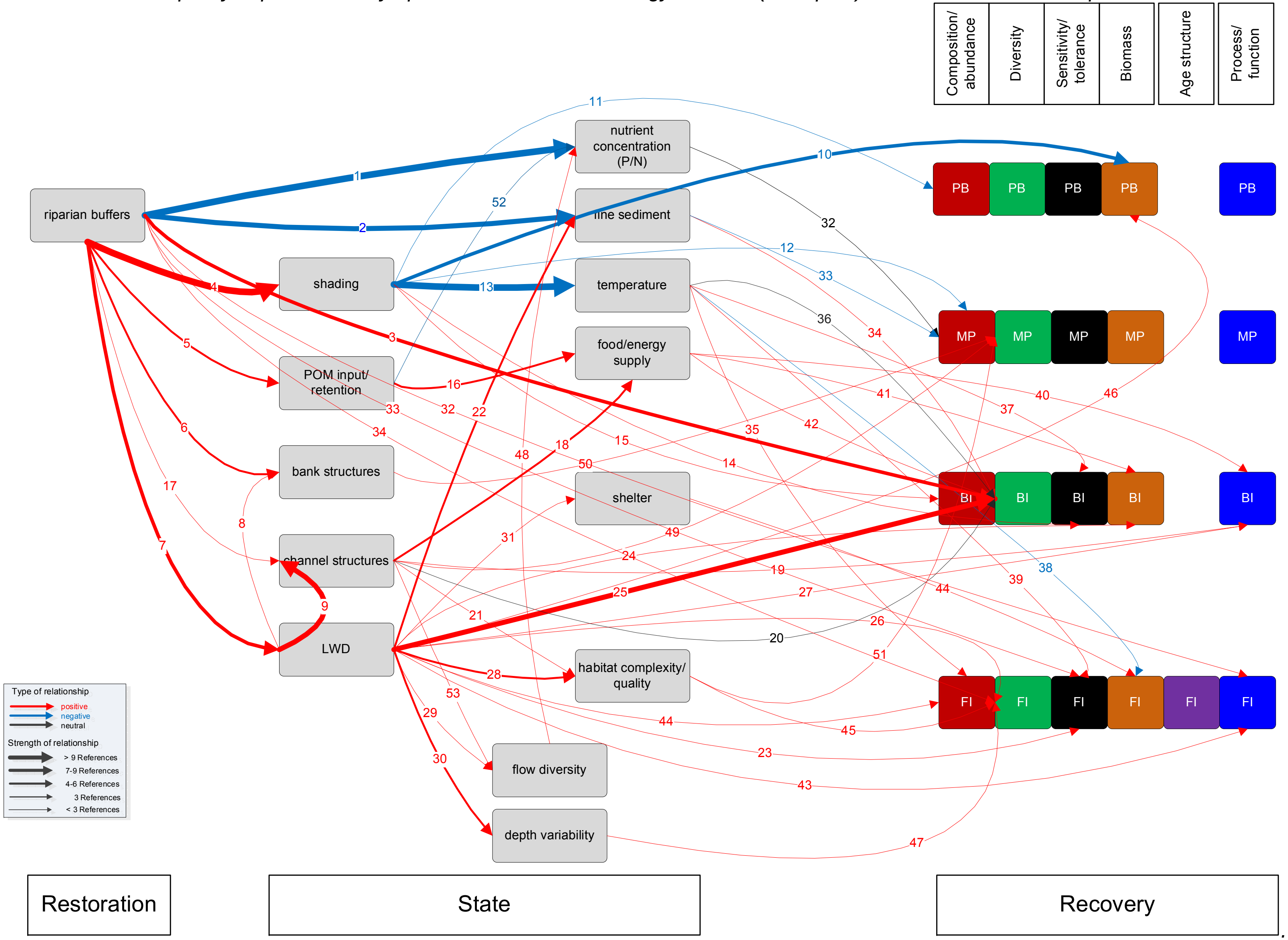
Annex 1 contains the three Conceptual Models developed in this study. Each example begins with an overview of the full model, followed by four models addressing each organism group: phytobenthos (PB), aquatic macrophytes (MP), benthic invertebrates (BI) and fish (FI).

Arrow thickness and colour are explained in Box 1 (legend).

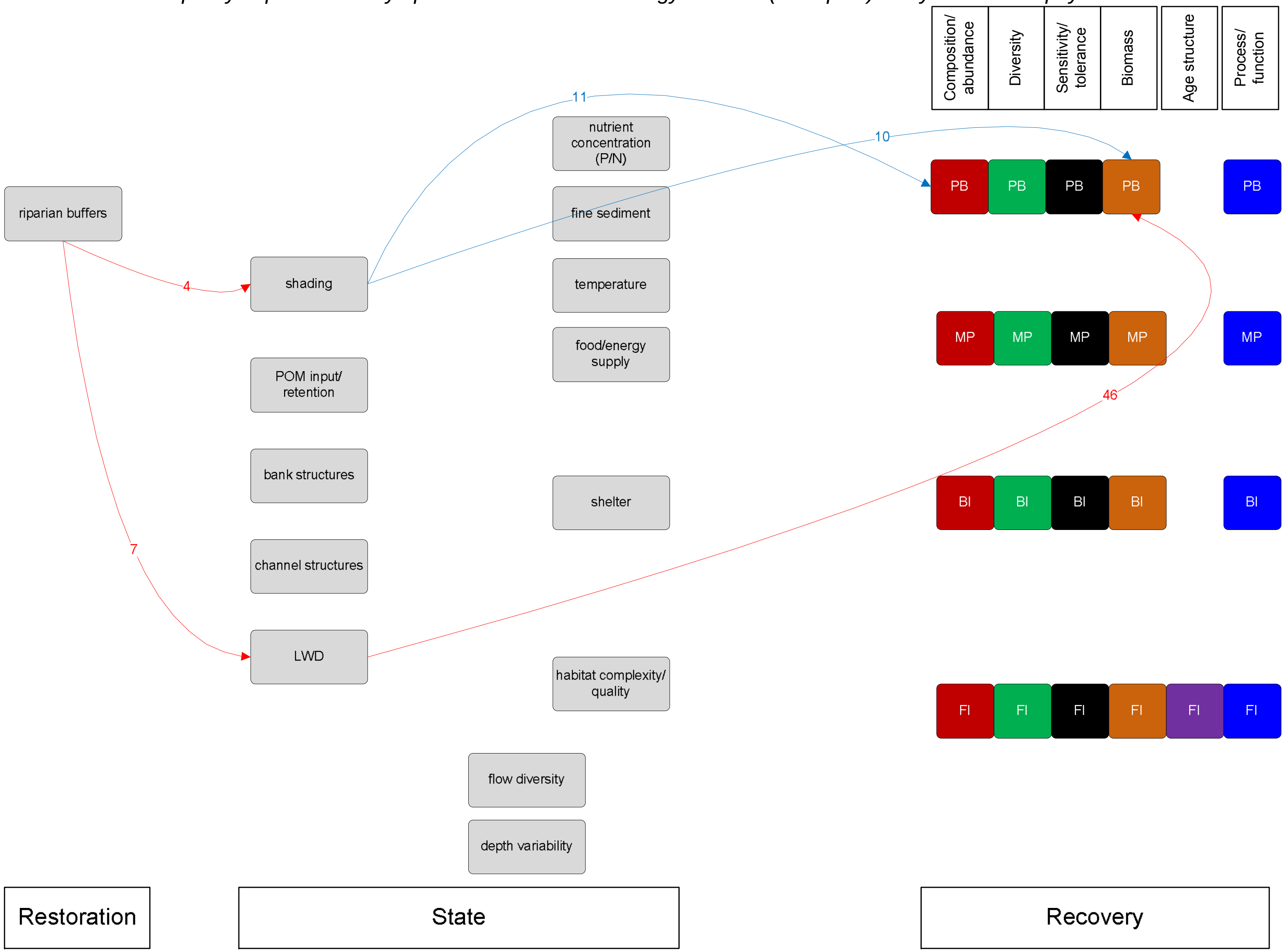


Box 1: Legend to the Conceptual Models.

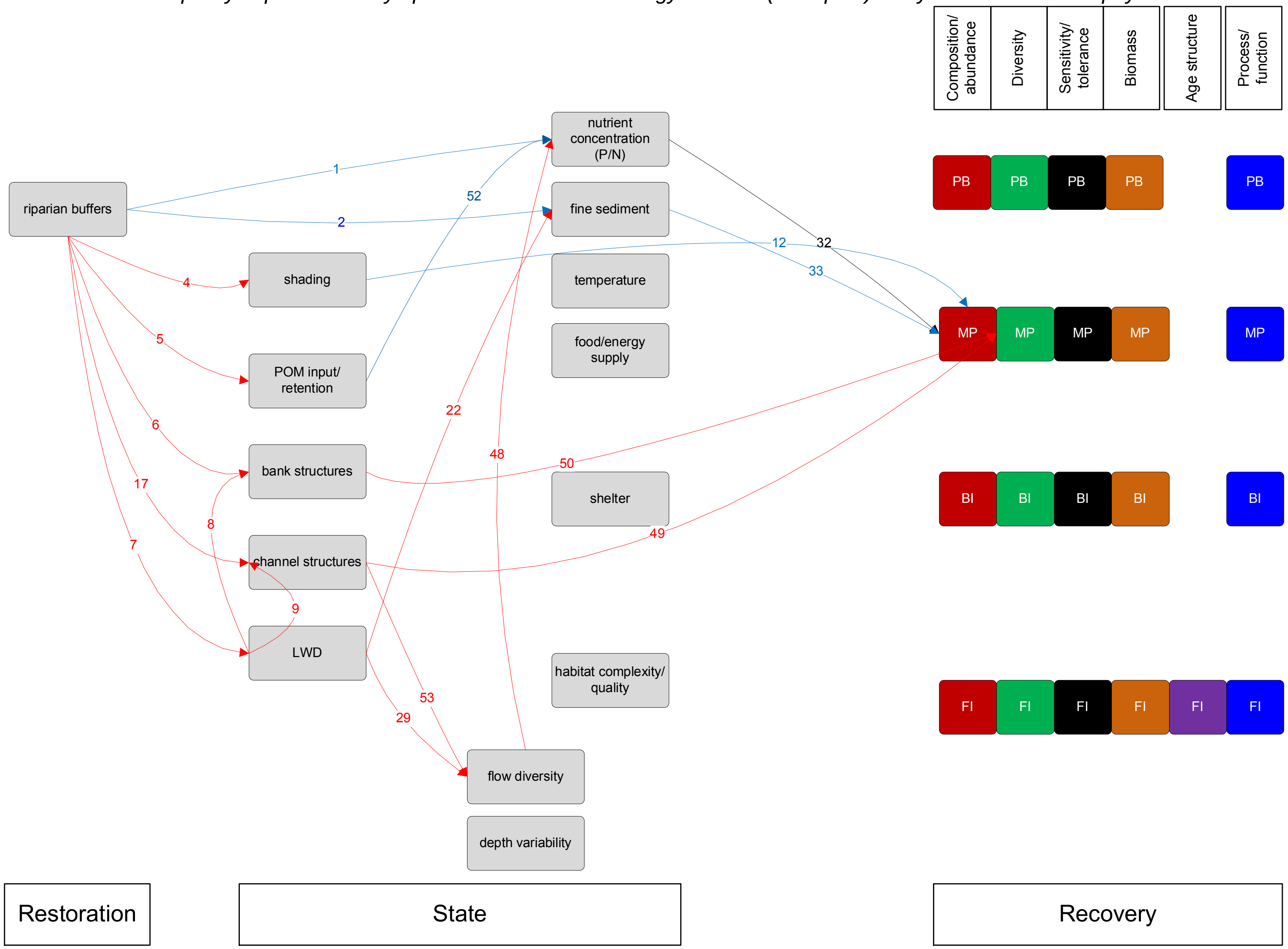
Annex 1.1: Water quality improvement by riparian buffers in low-energy streams (RLwqua1). Thickness of arrows equivalent to the number of references.



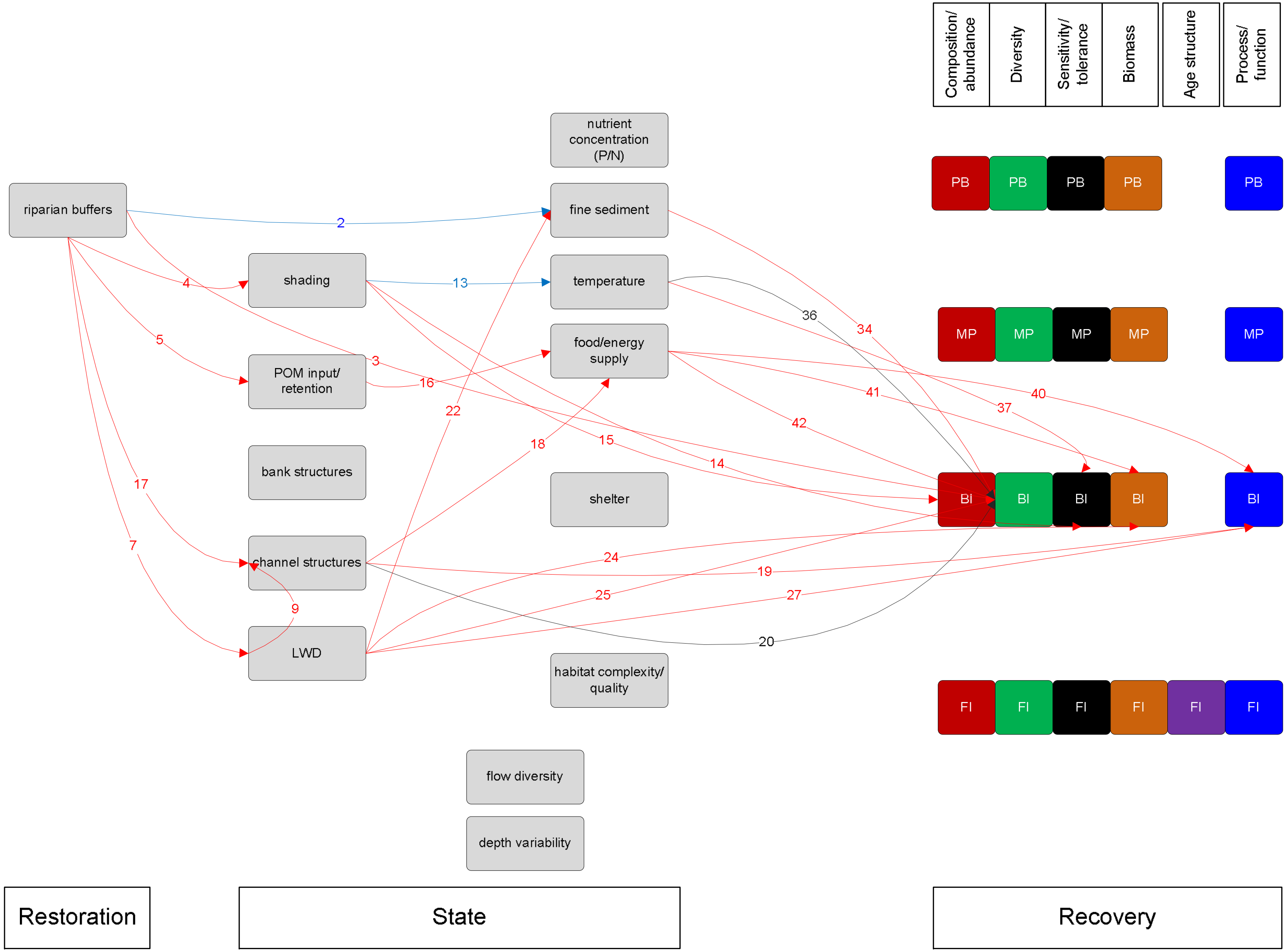
Annex 1.2: Water quality improvement by riparian buffers in low-energy streams (RLwqua1). Only links to the phytobenthos are shown.



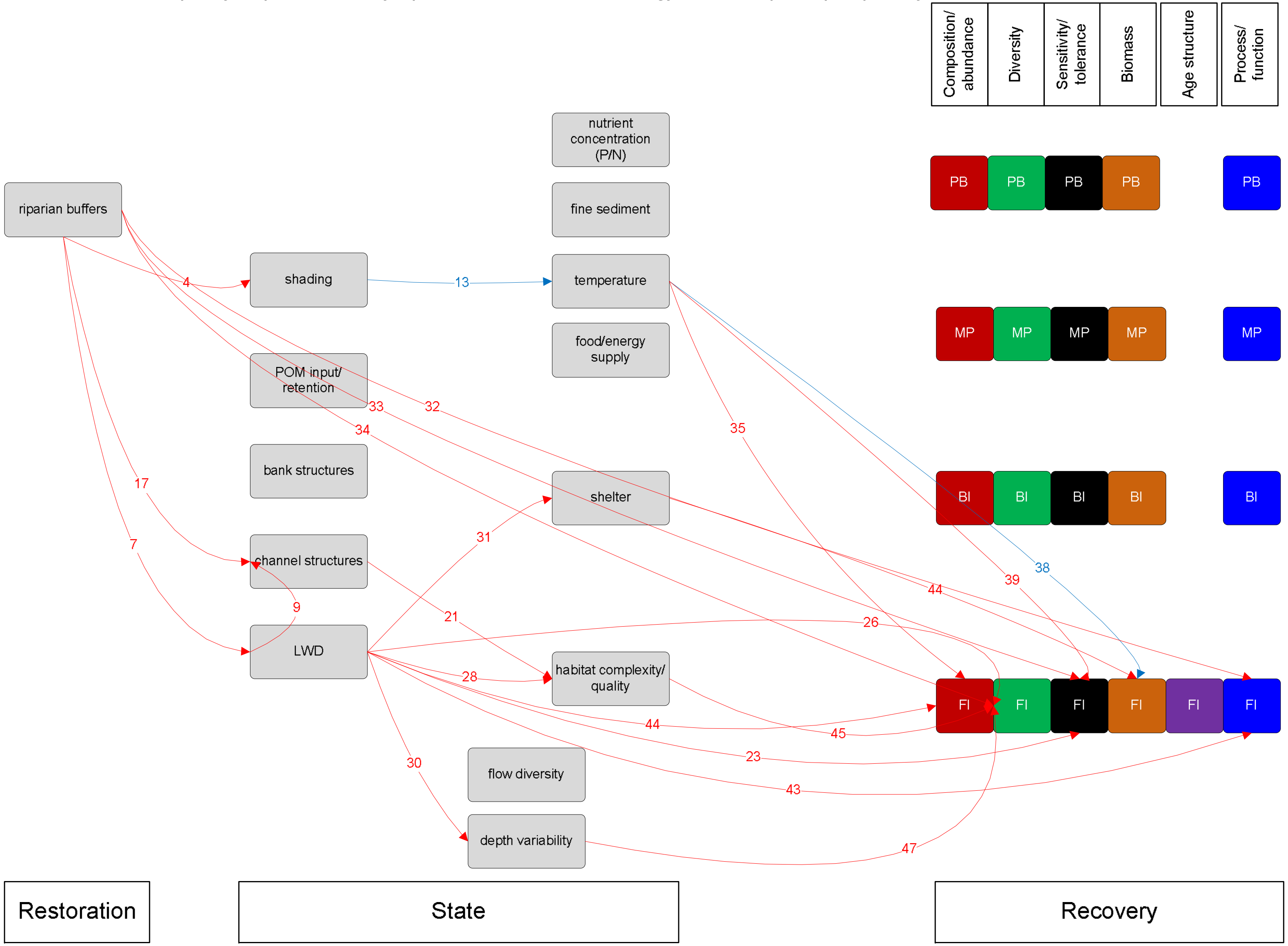
Annex 1.3: *Water quality improvement by riparian buffers in low-energy streams (RLwqua1). Only links to the macrophytes are shown.*



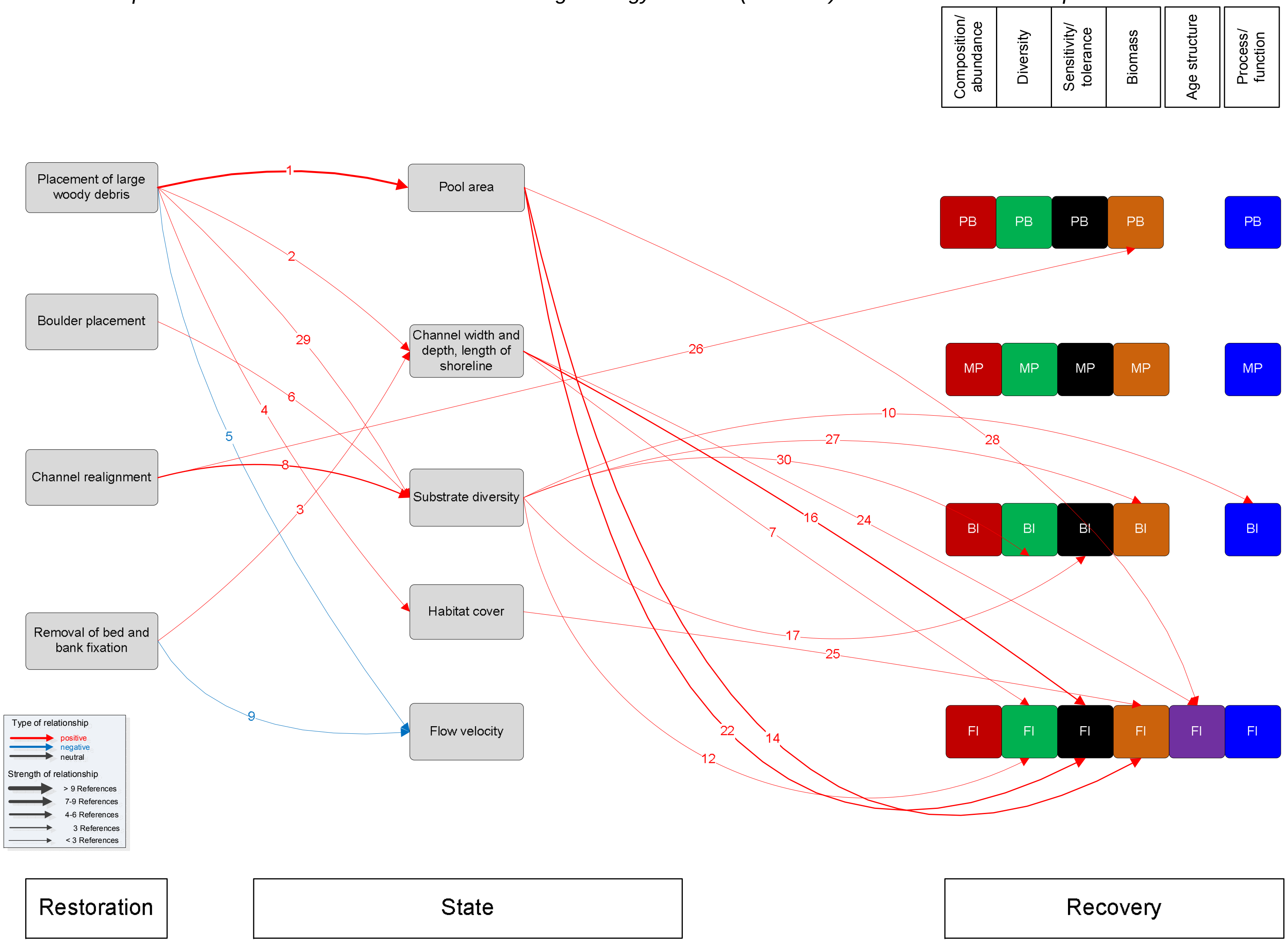
Annex 1.4: Water quality improvement by riparian buffers in low-energy streams (RLwqua1). Only links to the benthic macroinvertebrates are shown.



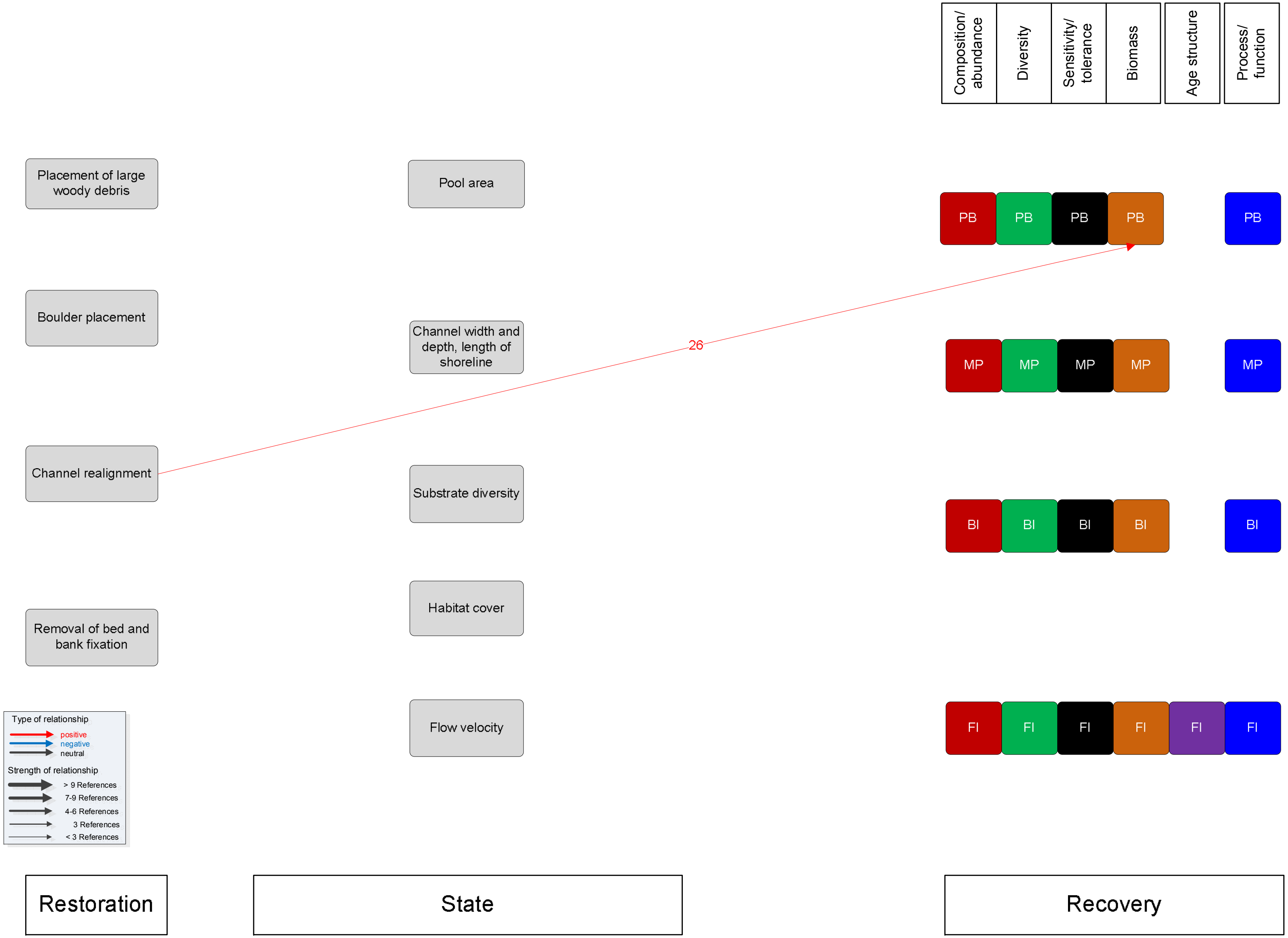
Annex 1.5: Water quality improvement by riparian buffers in low-energy streams (RLwqua1). Only links to the fish are shown.



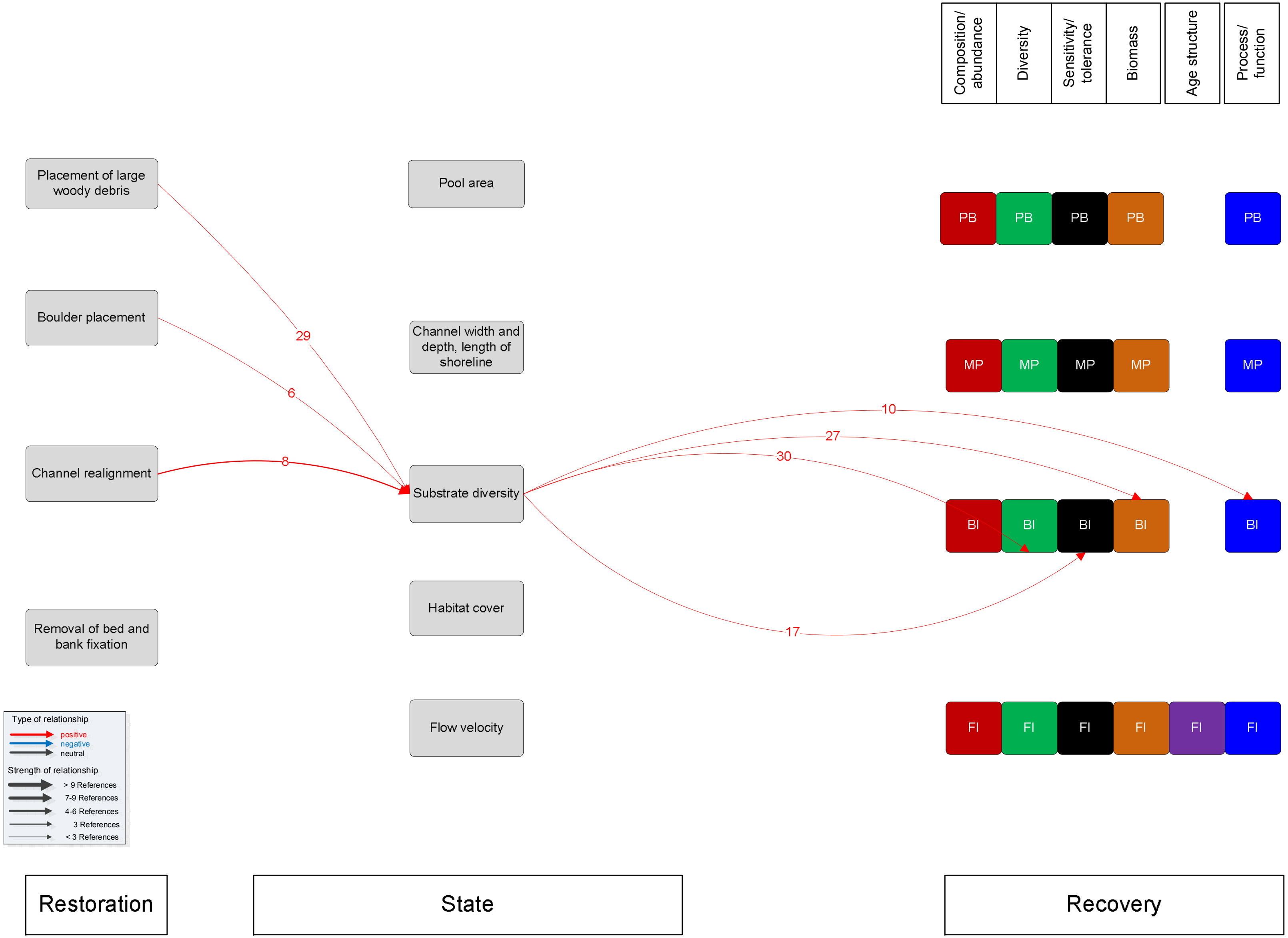
Annex 1.6: Improvement of in-stream habitat structure in high-energy streams (RHhabi1). Thickness of arrows equivalent to the number of references.



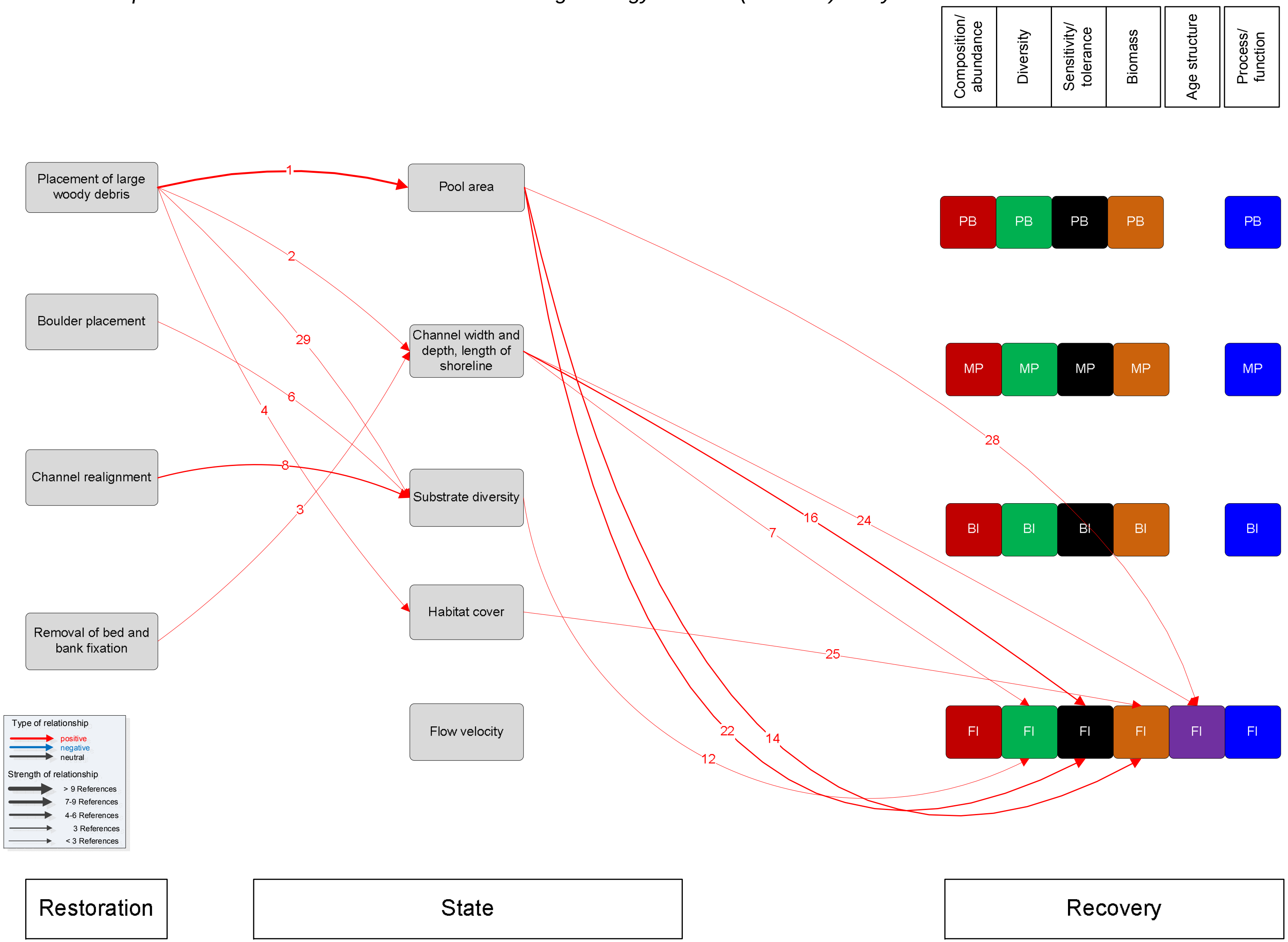
Annex 1.7: Improvement of in-stream habitat structure in high-energy streams (RHhabi1). Only links to the phytobenthos are shown.



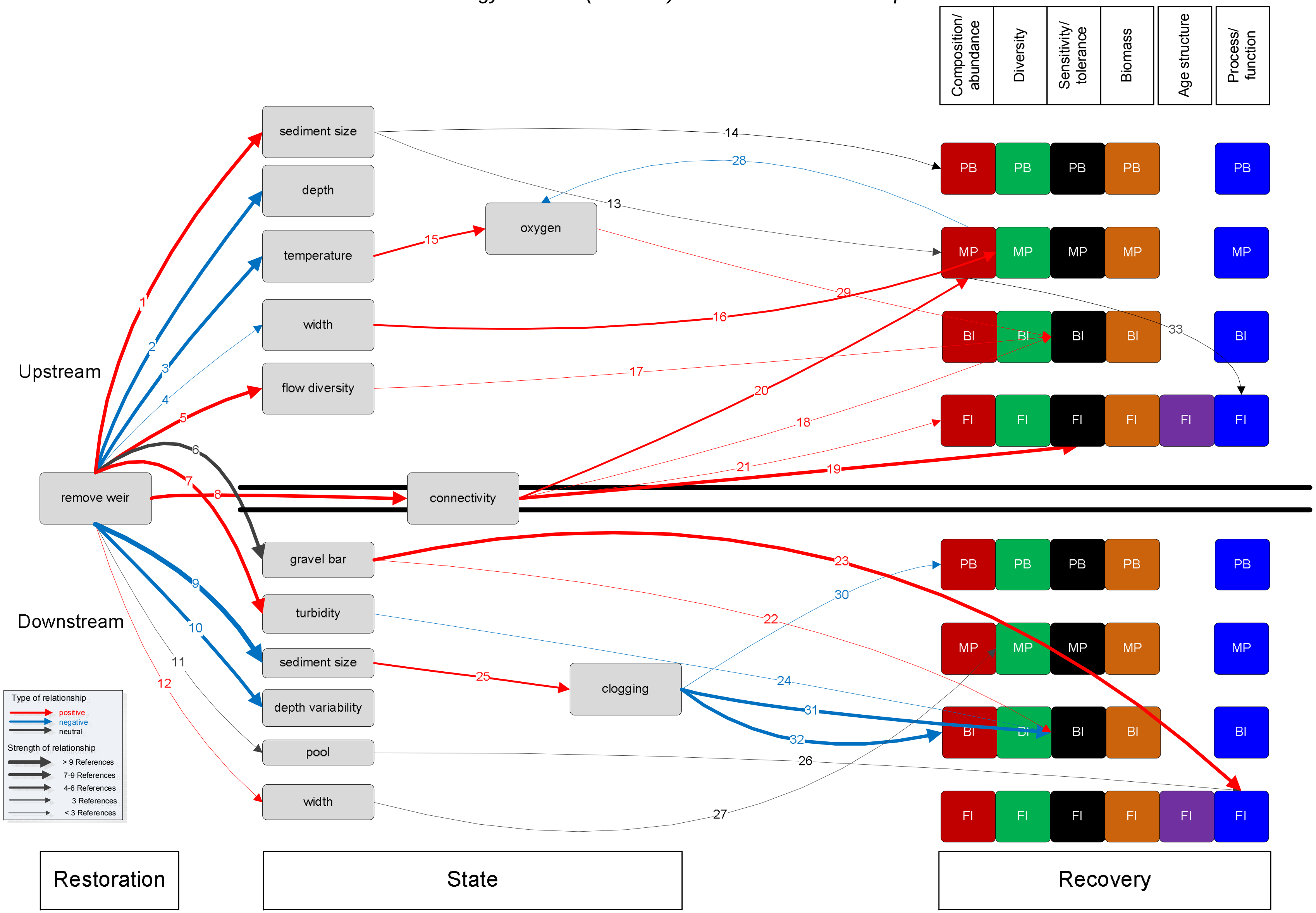
Annex 1.8: Improvement of in-stream habitat structure in high-energy streams (RHhabi1). Only links to the benthic invertebrates are shown.



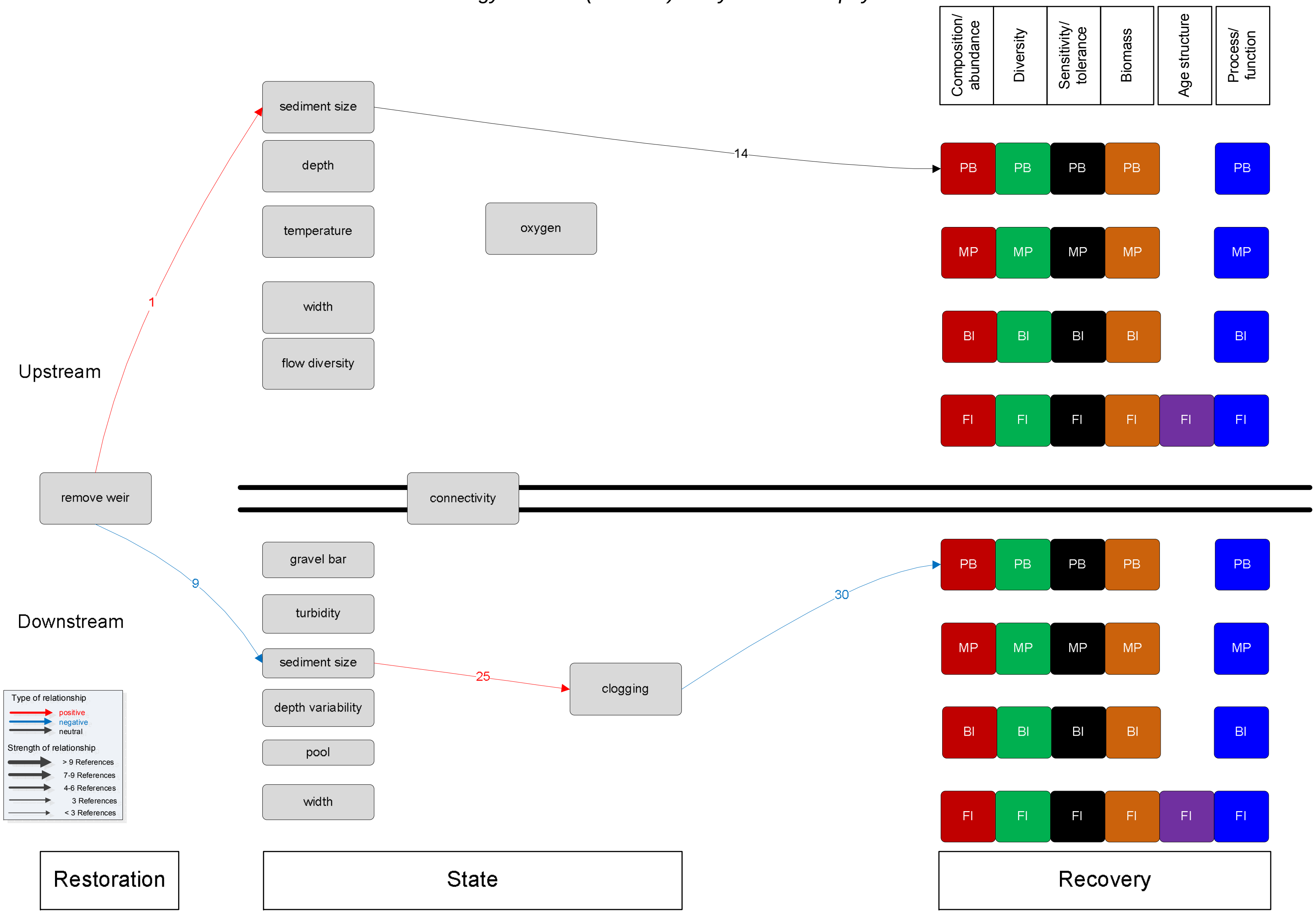
Annex 1.9: Improvement of in-stream habitat structure in high-energy streams (RHhabi1). Only links to the fish are shown.



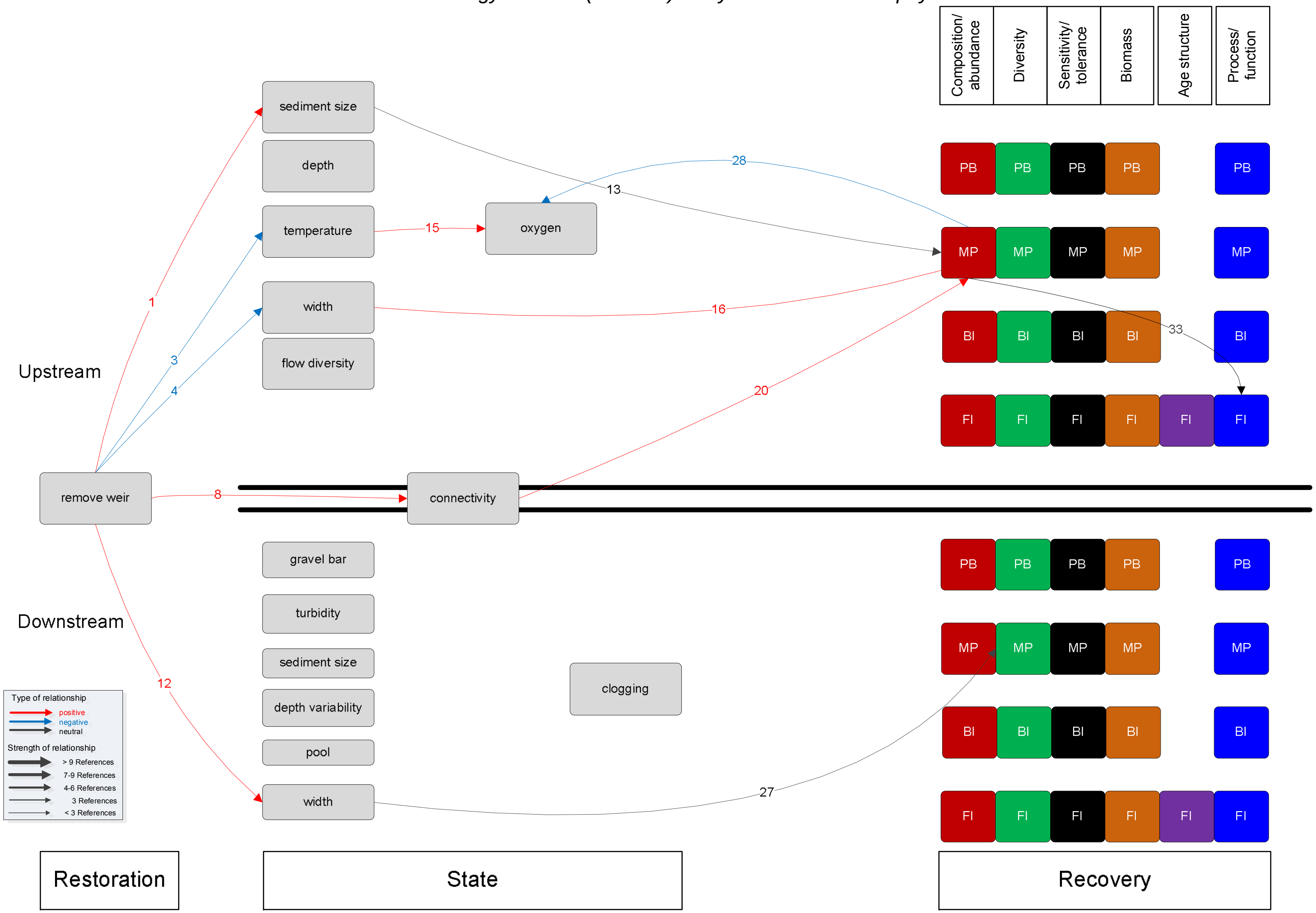
Annex 1.10: Removal of weirs and dams in low-energy streams (RLweir1). Thickness of arrows equivalent to the number of references.



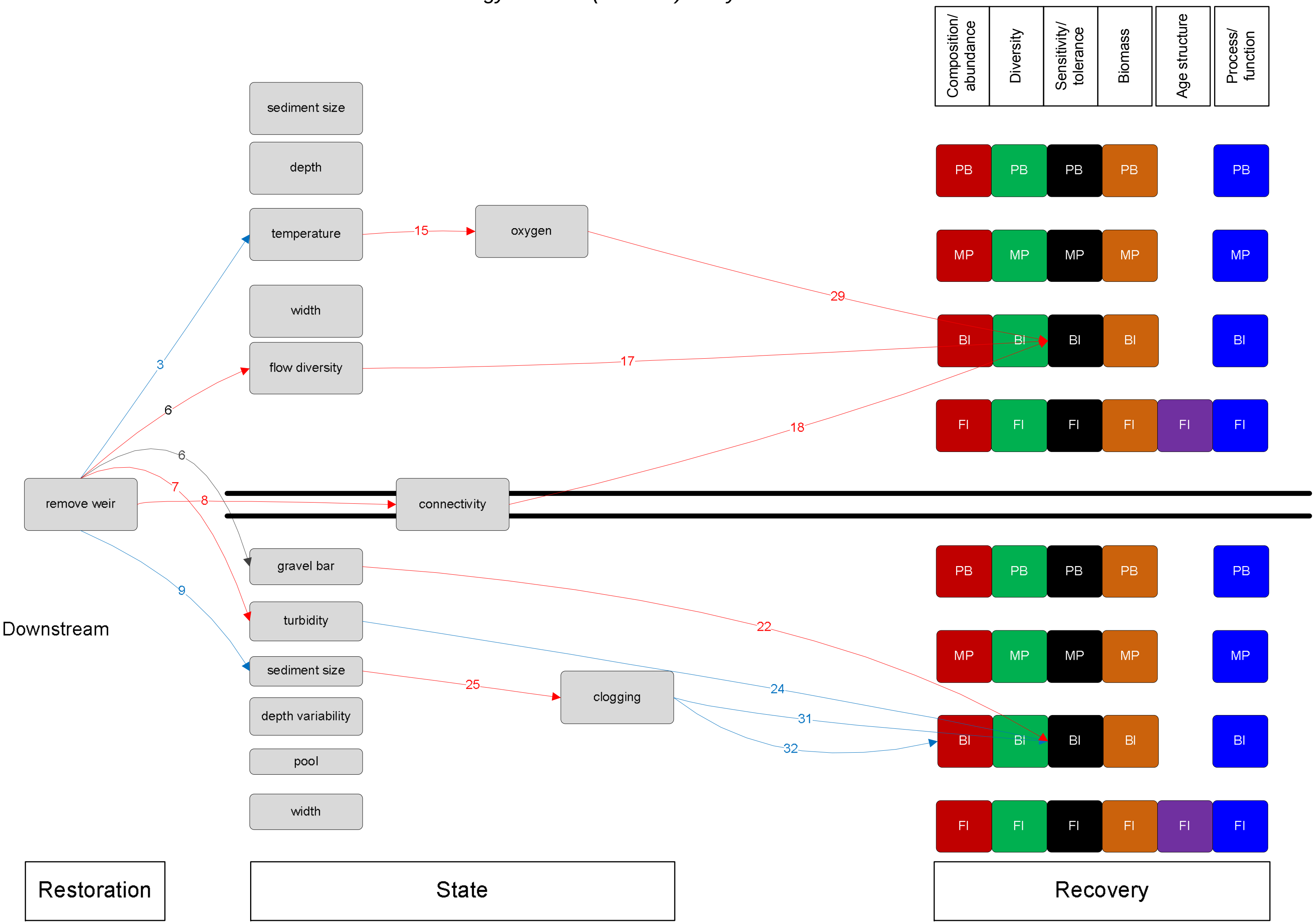
Annex 1.11: Removal of weirs and dams in low-energy streams (RLweir1). Only links to the phytobenthos are shown.



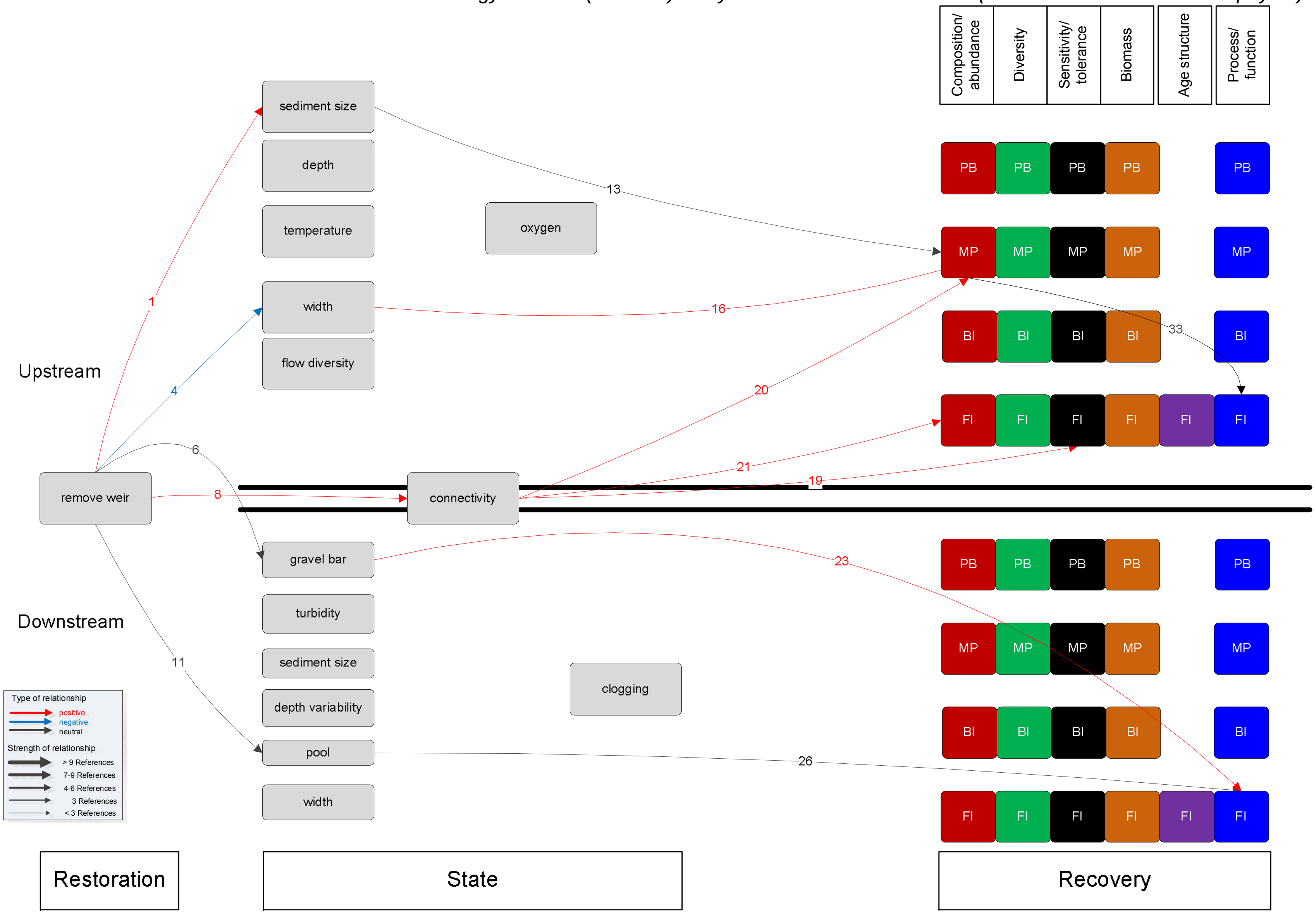
Annex 1.12: Removal of weirs and dams in low-energy streams (RLweir1). Only links to the macrophytes are shown.



Annex 1.13: Removal of weirs and dams in low-energy streams (RLweir1). Only links to the benthic macroinvertebrates are shown.



Annex 1.14: Removal of weirs and dams in low-energy streams (RLweir1). Only links to the fish are shown (and interactions with macrophytes).



Annex 2: Literature data sheets

The Excel sheets in Annex 2.1, 2.2 and 2.3 show the linkages in the three Conceptual Models presented in Annex 1 that are referred to by the reviewed body of literature. Full references can be found in the list of references on page 44ff.

Annex 2.1: Linkage numbers and references to the Conceptual Model on water quality improvement by riparian buffers in low-energy streams (RLwqua1).

A	B	C Model link No.																																																					D	E
Serial No.	Model code	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49	50	51	52	53	First author	Year
1	RLwqua1	x	x																																																				Dosskey MG	2001
2	RLwqua1	x	x																																																				Mankin KR	2007
3	RLwqua1				x						x																																												Davies-Colley RJ	1998
4	RLwqua1	x	x																																																				Schultz R	1995
5	RLwqua1			x	x									x																																									Broadmeadow S	2004
6	RLwqua1																																						x																Weatherley N	1990
7	RLwqua1	x									x	x	x	x	x																																								Parkyn SM	2003
8	RLwqua1. little effect study									x																x		x																											Larson M	2001
9	RLwqua1																									x																													Gerhard M	2000
10	RLwqua1									x													x						x		x																x		x							Brooks AP	2004
11	RLwqua1					x											x																									x	x												Wallace JB	1997
12	RLwqua1					x											x																										x												Northington RM	2006
13	RLwqua1								x	x									x				x		x	x	x		x	x	x																								Lester RE	2008
14	RLwqua1																		x	x		x																								x									Muotka T	2007
15	RLwqua1																									x											x	x																	Miller SW	2010
16	RLwqua1																		x																						x														Lepori F	2005
17	RLwqua1				x			x		x				x																																									Opperman JJ	2004
18	RLwqua1, no-effect example																				x																																		Harrison SS	2004
19	RLwqua1, no-effect example	x																																																					Sutton AJ	2009
20	RLwqua1																																																	x	x	x			Moustgaard-Pedersen TC	2006
21	RLwqua1																																																				x		Aldridge KT	2009
22	RLwqua1																																																x					x	Bukaveckas PA	2007
23	RLwqua1																									x																					x								Coe HJ	2009
24	RLwqua1																								x	x		x																											Entrekin SA	2009
25	RLwqua1																																											x	x										Cederholm CJ	1997
26	RLwqua1, almost no effect study									x																x																													Hilderbrand	1997
27	RLwqua1			x																															x																				Jowett	2009
28	RLwqua1		x	x	x		x	x			x																																												Quinn J	2009
29	RLwqua1	x																																																				Kaushal SS	2008
30	RLwqua1							x																																															McBride M	2008
31	RLwqua1, little effect study																			x																														x					Muotka T	2002
32	RLwqua1																															x	x	x																				Penczak T	1995
33	RLwqua1						x																																															Shields FD	1995
34	RLwqua1			x	x									x		x																																							Becker A	2009
35	RLwqua1			x														x				x																																	Pedraza GX	2008
G1	RLwqua1	x	x																																																				Correll DL	2005
G2	RLwqua1							x		x																																													Warren DR	2009
G3	RLwqua1																																				x																	Haidekker A	2007
G4	RLwqua1										x																																											Ghermandi A	2009
G5	RLwqua1								x	x													x						x	x	x																								Chen X	2008
G6	RLwqua1				x			x						x																																									Davies-Colley RJ	2009
G7	RLwqua1, contrasting example	x	x		x		x																																																Parkyn SM	2005
G8	RLwqua1, contrasting example	x	x	x	x									x																																									Castelle AJ	1994
G9	RLwqua1				x									x																						x				x															Barton DR	1985
G10	RLwqua1	x	x		x									x																																									Osborne LL	1993
G11	RLwqua1	x	x		x	x		x		x				x			x							x								x																							Wenger S	1999
G12	RLwqua1	x																																																					Fennessy MS	1997
G13	RLwqua1				x									x																						x				x															Whitledge GW	2006

Annex 2.2: Linkage numbers and references to the Conceptual Model on in-stream habitat improvement in high-energy streams (RHhabi1).

A	B	C Model link No.																														D	E
Serial No.	Model code	1	2	3	4	5	6	7	8	9	10		12		14		16	17					22		24	25	26	27	28	29	30	First author	Year
1	RHhabi1	x																														Hilderbrand RH	1997
2	RHhabi1								x		x							x														Herbst DB	2009
3	RHhabi1																													x	x	Gerhard M	2000
4	RHhabi1								x																		x	x				Moerke JE	2004
5	RHhabi1																									x						Binns NA	2004
6	RHhabi1			x			x	x		x			x		x		x								x							Jungwirth M	1995
7	RHhabi1	x																											x			Cederholm CJ	1997
8	RHhabi1																															Baldigo BP	2008
9	RHhabi1	x	x		x	x									x																	Riley SC	1995
10	RHhabi1																x															Muhar S	2008
11	RHhabi1	x																					x									Roni P	2001
12	RHhabi1														x								x									Roni P	2006

Annex 2.3: Linkage numbers and references to the Conceptual Model on weir removal in low-energy streams (RLweir1).

Serial No.	Model code	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	First author	Year
1	RLweir1																										x								Schlosser	1982
2	RLweir1																										x								Harvey	1991
3	RLweir1																			x		x													Iversen	1993
4	RLweir1			x		x										x																		x	Hill	1994
5	RLweir1	x																																	Kanehl	1997
6	RLweir1								x																										Poff	1997
7	RLweir1													x	x																				Baattrup-Pedersen	1999
8	RLweir1																											x							Kemp	1999
9	RLweir1																							x											Bednarek	2001
10	RLweir1		x	x		x					x					x														x		x			Bushaw-Newton	2002
11	RLweir1			x		x	x		x											x	x	x		x											Gregory	2002
12	RLweir1		x		x	x			x	x							x			x															Hart	2002
13	RLweir1																																		Pizzuto	2002
14	RLweir1				x												x				x							x							Shafroth	2002
15	RLweir1			x								x				x							x				x								Stanley	2002
16	RLweir1		x	x				x								x													x						Chaplin	2003
17	RLweir1	x					x	x		x														x		x									Hart	2003
18	RLweir1	x	x				x	x		x	x																								Randle	2003
19	RLweir1						x	x		x	x	x	x								x					x									Rathburn	2003
20	RLweir1	x								x																x						x	x		Pollard	2004
21	RLweir1																																		Doyle	2005
22	RLweir1								x											x															Schmitt	2005
23	RLweir1									x														x							x	x	x		Thomson	2005
24	RLweir1																x			x															Woolsey	2005
25	RLweir1																								x						x	x	x		Orr	2006
26	RLweir1							x		x	x																								Cheng	2007
27	RLweir1													x	x																				Kuhar	2007
28	RLweir1																			x															Leaniz	2008
29	RLweir1																	x	x																Maloney	2008
30	RLweir1		x		x						x	x	x																						Burroughs	2009
31	RLweir1										x												x									x	x		Tzsydel	2009