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Ecological indicators for stream restoration success

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ABSTRACT

Exploitation of freshwater resources is essential for sustenance of human existence and alteration of rivers, lakes and wetlands has facilitated economic development for centuries. Consequently, freshwater biodiversity is critically threatened, with stream ecosystems being the most heavily affected. To improve the status of freshwater habitats, e.g. in the context of the European Water Framework Directive and the US Clean Water Act, it is essential to implement the most effective restoration measures and identify the most suitable indicators for restoration success. Herein, several active and passive bioindication approaches are reviewed in light of existing legal frameworks, current targets and applicable implementation of river restoration. Such approaches should move from the use of single biological indicators to more holistic ecological indicators simultaneously addressing communities, multiple life stages and habitat properties such as water quality, substrate composition and stream channel morphology. The proposed Proceeding Chain of Restoration (PCoR) can enable the integration of natural scientific, political and socioeconomic dimensions for restoration of aquatic ecosystems and associated services. Generally, an analysis that combines target species-based active bioindication with community-based passive bioindication and multivariate statistics seems to be most suitable for a holistic evaluation of restoration success, as well as for the monitoring of stream ecosystem health. Since the response of biological communities to changing environmental conditions can differ between taxonomic groups and rivers, assessments at the ecosystem scale should include several levels of biological organisation. A stepwise evaluation of the primary factors inducing disturbance or degradation is needed to integrate increasing levels of complexity from water quality assessments to the evaluation of ecological function. The proposed PCoR can provide a step-by-step guide for restoration ecologists, comprising all planning steps from the determination of the conservation objectives to the use of ecological indicators in post-restoration monitoring.

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1. Introduction

Freshwater ecosystems are hotspots for biodiversity (Strayer and Dudgeon, 2010; Geist, 2011), containing 6–10% of all species and one-third of all vertebrate species worldwide (Dudgeon et al., 2006; Balian et al., 2008). Freshwater ecosystems are strongly influenced by human settlements, and agricultural or industrial land use (Convention on Biological Diversity, CBD, United Nations, 1992; Tockner et al., 2009; Feld et al., 2011), making them particularly prone to degradation. Freshwater resources are essential for sustaining human existence and the alteration of rivers, lakes and wetlands has tracted the economic development for centuries (Sala et al., 2000). This is most evident in central European countries with their typically high population density and early industrialisation (Hladyz et al., 2011a). In other parts of the world, there is also high pressure on freshwater systems due to overfishing (e.g. Mekong, Asia, Kang et al., 2009; Lake Viktoria, Afrika, Payne, 1976), intensive agriculture in floodplains (e.g. Piracicaba River, South America, Filoso et al., 2003; Nile, North Africa, Wahaab and Badawy, 2004; Betsiboka River in Madagascar, Raharimahefa and Kusky, 2010), chemical pollution (e.g. Amazon in Suriname, South America, Mol et al., 2001; Pilcomayo River in South America, Smolders et al., 2003; Liaohe River, China, Zhang et al., 2010) and structural degradation (e.g. Yangtze River, China, Fu et al., 2003; Rio Paranà, South America, Sanches et al., 2006; Mekong, Asia, Kang et al., 2009). Consequently, freshwater biodiversity is critically threatened worldwide (Ricciardi and Rasmussen, 1999; Jenkins, 2003) with stream ecosystems being most heavily affected (Stein and Flack, 1997; Pimm et al., 2001; Gleick, 2003). In recent years, there has been growing awareness that restoration of freshwater habitats is essential to maintain ecosystem services, especially food and drinking water supply (Millennium Ecosystem Assessment, 2005). The need for aquatic ecosystem restoration is now well established, particularly in Europe and North America (Kondolf et al., 2007). Biodiversity conservation objectives, along with ethical and recreational reasons, have resulted in numerous efforts to restore freshwater ecosystems. Over the last 30 years, river restoration has become widely applied (Bernhardt et al., 2007; Bernhardt and Palmer, 2011) and is likely to play a major role in environmental management and policy decisions in the future (Palmer et al., 2004). The financial resources invested in river restoration in the United States since 1990 were estimated to exceed one billion dollars per year (Bernhardt et al., 2005), underlining the economic importance. Regardless of the immense financial effect and the numerous restoration projects, surprisingly little is known about the factors that drive successful restoration (Palmer et al., 2005; Feld et al., 2011). Herein, the term "restoration success" refers to the criteria of measurably improving flow dynamics, ecological conditions, river health, ecosystem self-sustainability, and resilience to external perturbations, as proposed in Palmer et al. (2005). Furthermore, implemented measures must produce no lasting harm (first stated by Leopold, 1948). Also, it has been increasingly recognised that freshwater ecosystem restoration depends on the combination of scientific, technological, social, economic, and political efforts. Among these efforts, scientific assessment of freshwater ecosystem health provides the foundation for political decisions and legislation focusing on indicators of stream restoration success.

This review aims to contribute to the international exchange and collaboration among scientists, engineers, economists and politicians. Specifically, the importance of policy and legal frameworks for river restoration is addressed using examples from Europe and North America. Moreover, the complex interactions of ecological, technical and socio-economic factors determining restoration success are reviewed. Different active and passive bioindication methodologies are discussed in light of legal frameworks, current targets, and applicability in river restoration implementation. We propose the Proceeding Chain of Restoration (PCoR), a holistic approach which integrates the standardised monitoring of ecological indicators, management practices and structures, and which is useful in moving current restoration evaluation from single species-based bioindication to holistic community-based bioindication and assessment.

2. European legislation linked to stream ecosystem restoration

The threat of extinction of many aquatic organisms and loss of aquatic biodiversity has found its way into the consciousness of policy makers and politicians in many countries. This is reflected in the solid fundament of legislation, rules and regulations that protect aquatic species and their habitats (Table 1). Some of these were only recently established.

The European Union reacted to the predicted loss of biodiversity and resulting decrease in human well-being by proclaiming the Water Framework Directive (WFD) in 2000 (European Parliament, 2000). The purpose of the WFD is to enforce a Europe-wide implementation of a comprehensive approach to protect the sustainability of water resources (Hering et al., 2010) by integrating both chemical (concentrations of pollutants and nutrients, European Parliament, 2000) and biological quality components (fishes, macroinvertebrates, macrophytes, phytobenthos/phytoplankton, European Parliament, 2000). The primary goal of the directive is that all major surface water (natural surface water bodies, heavily modified water bodies and artificial water bodies) and groundwater systems reach "good ecological status" or "good ecological potential" by 2015.

Within this framework, every European member state was required to survey the status of surface waters as a first step. River Basin Management Plans had to be developed as subsequent step by 2009. These documents are water policy plans specific to catchment areas and drainage systems, which outline a strategic level of planning (WFD, European Parliament, 2000 Annex VII). They are legally binding for all federal authorities and contain a program of measures (PoMs, WFD, European Parliament, 2000 Annex VI), which can serve as a toolbox to reach WFD goals.

All member states were obligated to include the WFD into national legislation. In Germany, this was accomplished by revising the Water Management Act of 1957 (Wasserhaushaltsgesetz, WHG) in 2010 (Bundesministerium der Justiz, 2009a, implemented as WHG 2010). The revision now strongly emphasises the ecological functionality of rivers and streams.

Besides the WFD, several European directives and national equivalents (Table 1) related to the topic of river ecological function were implemented in the last decade. For example, directives to regulate discharges of certain hazardous substances (WPD, European Parliament, 2006) and the Floods Directive (FLD, European Parliament, 2007) both contain instructions regarding aspects of river health. All of these water-related legal regulations create an urgent need for swift successful restoration of running waters.

3. Stream restoration complexity

River restoration can be influenced by ecological, technical and socio-economic factors which all interact in complex ways (Fig. 1). Consequently, it may be difficult to find the "silver bullet" solution to determine restoration success. Many factors affect restoration activities, but their impact is strongly correlated with the complexity of the restoration target (Feld et al., 2011). On an ecosystem scale, there are naturally complex interactions within species, between species, and between species and habitats, all based on the different sensitivities of species or life stages to

Table 1
European directives and national regulations of Germany and USA which contribute as labour contracts to the restoration of river ecological function. The abbreviations: W = improvement of the water quality, H = protection and improvement of habitat quality and S = species protection, illustrate the main focus of which the European directive refers to the German implementation and North American regulations analogously.

European directive	Code/year	W	H	S	Content	National regulations (examples)	
						Germany	USA
Water Framework Directive	WFD 2000/60/EC	X	X	X	Protection, restoration and long-term sustainable use of clean water	Wasserhaushaltsgesetz (WHG) 2010	Clean Water Act (CWA) 1972
The Habitats Directive (Natura 2000)	FFH 92/43/EEC		X	X	Maintenance of biodiversity, accounting of economic, social, cultural and regional requirements	Bundesnaturschutzgesetz (BNatSchG) 2009b	Endangered Species Act (ESA) 1973
Environmental Impact Assessment Directive	EIA 85/337/EEC	X	X	X	Integration of environmental considerations into the preparation of projects, plans and programs to reduce their environmental impact	Gesetz über die Umwelt-verträglichkeitsprüfung (UVPG) 2010; Baugesetzbuch (BauGB) 2011a	National Environmental Policy Act (NEPA) 1969
Urban Waste Water Directive	WWD 91/271/EEC	X			Protection of the environment from the adverse effects of urban waste water discharges and discharges from certain industrial sectors	Abwasserverordnung (AbwV) 2009c	Clean Water Act (CWA) 1972
Water Pollution by Discharges of Certain Dangerous Substances Directive	WPD 2006/11/EC	X			Regulation of potential aquatic pollution by chemicals, including discharges to inland surface waters, territorial waters, inland coastal waters and ground water	WHG 2010; AbwV 2009c	Toxic Substances Control Act (TSCA) 1976
Integrated Pollution Prevention and Control Directive	IPPC 2008/1/EC	X			Regulation of six categories of industrial activities: energy industries, production and processing of metals, mineral industries, chemical industries, waste management and other activities	Bundes-Immissionsschutzgesetz (BImSchG) 2011c; Kreislaufwirtschafts- und Abfallgesetz (Krw-/AbfG) 2011b; WHG 2010; AbwV 2009c	Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) 1996; Toxic Substances Control Act (TSCA) 1976; Clean Water Act (CWA) 1972
Nitrates Directive	NID 91/676/EEC	X			Prevention of nitrate contamination from agricultural sources concerning ground and surface waters	Düngeverordnung (DüV) 2007	No nitrate specific regulation besides CWA
Drinking Water Directive	DWD 98/83/EC	X			Consumer health protection and ensurance of wholesome and clean water	Trinkwasserverordnung (TrinkwV) 2011d	Safe Drinking Water Act (SDWA) 1974; Public Health Security and Bioterrorism Preparedness and Response Act (Bioterrorism Act) 2002
Floods Directive	FLD 2007/60/EC		X		Reduction and management of flood risks to human health, the environment, cultural heritage and economic activity	Gesetz zur Verbesserung des vor-beugenden Hochwasserschutzes (Hochwasserschutzgesetz) 2005	Flood Control Act (FCA) 1965
European Red List (European Commission, 2011)	ERL IUCN 2010 ERL			X	Detailed and up to date information on bio-diversity and conservation status of species	Rote Liste gefährdeter Tiere Deutschlands (Bundesamt für Naturschutz, 2011); BNatSchG 2009b	Endangered Species Act (ESA) 1973; IUCN Red List of North American Threatened Species
Public Access to Environmental Information	PAII 2003/4/EC				Establishment of a general right of any person to environmental information held by public authorities	Umweltinformationsgesetz (UIG) 2004	Emergency Preparedness and Community Right-to-know Act (EPCRA) 1986

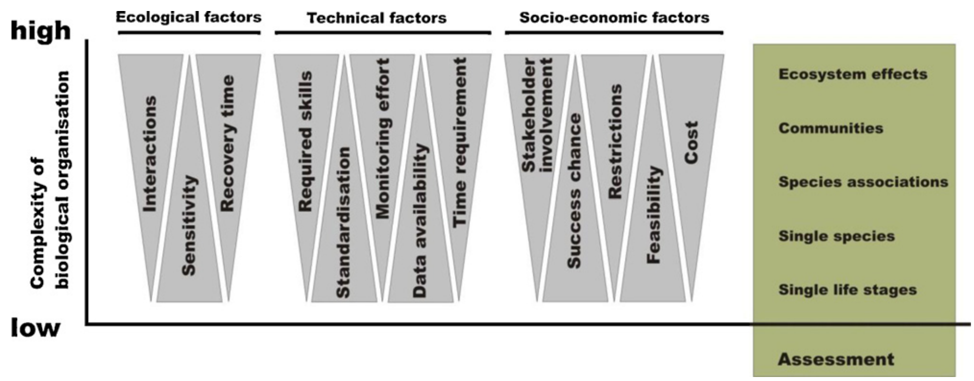

Fig. 1. River restoration is affected by ecological, technical and socio-economic factors which are displayed as grey triangles. The impact of the factor increases or decreases depending on the complexity of biological organisation of the restoration. The respective assessment scales for restoration targets of increasing biological complexity are highlighted in green.

associated ecosystem processes (Clarke and Warwick, 2001). Processes at the ecosystem level naturally need time to develop important ecosystem functions to result in a resilient and self-sustainable system. With increasing spatial scale and ecological complexity, different factors become crucial for the implementation of restoration projects. For instance, the restoration of complex ecosystem processes is strongly linked to extended recovery time (Power, 1999). Restoration at increasingly large spatial scales can lead to rising numbers of restrictions (e.g. agricultural land use, infrastructure, disposal systems and utility services, Hladyz et al., 2011a) which can limit the feasibility of restoration projects. These restrictions can intensify stakeholder involvement (Wohl et al., 2005), increase the number of required skills, and increase needed monitoring efforts (Manning et al., 2006). All of these factors contribute to rising project costs. Consequently, restoration at the ecosystem scale has low feasibility which in turn can lead to reduced political awareness. In contrast, the restoration of critical life stages or habitats for single species is often less complex and the recovery times at the population level can be short (Power, 1999). Additionally, easier technical implementation of the measure and high feasibility follow comparatively low costs. Limited funding can affect the post-restoration monitoring much more than the implementation of the restoration itself (Minns et al., 1996), causing insufficient investigation of river restoration success rates. Both the lack of standardised methods for cost-effective monitoring and the assessment of restoration measures may exacerbate the problem.

The high complexity of river restoration goals and measures, like the improvement of recreation, flood protection or ecological status, makes it difficult to focus on a single universal parameter (Palmer et al., 2005; Bernhardt et al., 2007). Consequently, the success of restoration is difficult to measure. For each intention, scales of success differ with sometimes contrary definitions (Jähnig et al., 2011). In particular, success in terms of increased recreational value or improved flood protection is not necessarily correlated with the improvement of river ecological function for aquatic species. While better recreation possibilities can be measured by counting visitor numbers, for example, and the success of flood protection by the reduction in flood damage, the improvement of ecological integrity is more complex and requires inclusion of multidimensional processes and interactions.

4. Stream restoration monitoring

Irrespective of the complexity of restoration projects, intensive monitoring enriches the knowledge about the drivers of successful restoration. However, to date, stream restoration is often based on trial and error, without effective monitoring of ecological improvements. For instance, in the last two decades, water authorities in Bavaria (one of the 16 German states) alone spent more than 300 million Euros for the implementation of river restoration projects (Table 2). The intentions of these projects varied from flood protection, compliance with national legislation, and improvement of recreational values, to the restoration of the ecological functionality of rivers. A survey of the official web-pages from Bavarian water authorities revealed that no data are available to determine the failure or success of the implemented measures for 86% of the projects due to the lack of any efficiency controls (Table 2). Only 10% of the small scale restoration projects (less than 1 km restored river or bank length, Bernhardt et al., 2005) or point restoration measures) reported in Table 2 were monitored on a short-term basis. None of these projects were monitored for more than one year. Long-term monitoring (exceeding one year) was carried out for only 7% of the large-scale restorations considered. Large-scale measures (more than 1 km restored river or bank length) seem to be monitored more intensively than small scale measures. However, with only 25% large-scale projects monitored in general, these efforts are still insufficient. The monitoring program for only 4% of all restorations included investigations of the pre-restoration status which is an important reference for holistically evaluating project success (Palmer et al., 2005; Hering et al., 2010; Friberg et al., 2011). This example illustrates the deficits in restoration monitoring even in well-developed countries.

Without a systematic and target-oriented evaluation of restoration measures, it is not feasible to detect the factors predominantly driving restoration success (Feld et al., 2011). Consequently, it is difficult to identify the most effective restoration strategy for future projects. Generally, biological indicators are useful for monitoring the effect of changing environmental conditions on surface waters (Bellinger and Sigee, 2010). However, assessment protocols like the WFD in Europe or the Rapid Bioassessment in North America to date are not fully suitable for detection of restoration success which can vary highly in type and scale (Sundermann et al., 2011).

5. The role of bioindication in river restoration

A particular species, or group of species, whose function, population, or status can be used to determine ecosystem or environmental changes can act as biological indicator (Dziock et al., 2006). The response of the indicator organisms can differ in sensitivity, ranging from changes in physiology, behaviour and morphology to changes in survival and mortality. Based on these responses, conclusions can be drawn about the ecological integrity of an ecosystem or the influence of potential stressors (Knoben et al., 1995; Dziock et al., 2006). While chemical and physical measurements of numerous variables can be expensive for monitoring

Table 2
Assessment of river restoration measures in Bavaria, Germany from 1994 to 2011. This investigation includes all restoration measures that can be found on the official web-pages of the Bavarian water authorities. Additional information about the drivers, the restoration goals and the monitoring and financial efforts were also drawn from the web. Restoration measures were classified as large-scale and small-scale measures according to the length of the restored river section. Large-scale restoration measures refer to all measures with more than 1 km restored bank length or river section (Bernhardt et al., 2005). All other restoration measures were considered as small-scale restoration.

	Total number of restorations	Large scale restorations	Small scale restorations
Number	101 [100%]	28 [100%]	73 [100%]
Monitoring			
No	87 [86%]	21 [75%]	66 [90%]
Short term	12 [12%]	5 [18%]	7 [10%]
Long term	2 [2%]	2 [7%]	0
Pre-restoration status	4 [4%]	2 [7%]	2 [3%]
Single group study	8 [8%]	2 [7%]	6 [8%]
Multi group study	6 [6%]	5 [18%]	1 [1%]
Restoration goal			
Fish passage	52 [51%]	11 [39%]	41 [56%]
Structural improvements	65 [64%]	25 [89%]	40 [55%]
Overall ecosystem health	22 [22%]	16 [57%]	6 [8%]
Drivers			
Flood protection and security	28 [28%]	11 [39%]	17 [23%]
Legislation	25 [25%]	4 [14%]	21 [29%]
Recreation	3 [3%]	2 [7%]	1 [1%]
Ecology	45 [45%]	11 [39%]	34 [47%]
Financial effort per measure in €	3.209.660 [$n = 50$]	12.204.524 [$n = 21$]	113.068 [$n = 29$]
[Range]	[11.000–146.000.000]	[153.000–146.000.000]	[11.000–1.800.000]

long-term ecosystem changes (Metcalfe-Smith, 1996), bioindication can provide easy and cost-effective tools for short- and long-term monitoring of environmental and ecosystem integrity (Neumann et al., 2003). This can be particularly important in monitoring restoration success when funding is typically limited.

To evaluate restoration success, suitable and effective indicators must fulfil general criteria including economic and logistic suitability and biological efficiency (McGeoch, 1998). Indicators should exhibit a narrow ecological range, rapid response to environmental change, well-defined taxonomy for reliable identification, wide distribution, and be inexpensive to sample (Bellinger and Sigee, 2010). Currently, fish, invertebrates, macrophytes, and algae are commonly used to monitor the status of freshwater ecosystems (Friberg et al., 2011). Many indicator systems and protocols have been developed for determining the condition of aquatic systems and for the evaluation of human impacts on these (Table 3). However, these evaluation systems can be used only in some cases to determine the success of small-scale restoration. None of them are practicable for detecting improvements of target species-oriented restoration, where species or even life stage-specific requirements and distinctive abiotic habitat variables have to be evaluated together to determine success. This can only be achieved by actively exposing the target species to ambient conditions at the restored sites (i.e. active bioindication, Schubert, 1991).

5.1. Aquatic indicator organisms for bioindication

Active bioindication can standardise monitoring methodology with respect to exposure duration, organism size, age classes or the general quality of indicator organisms (Knoben et al., 1995). Typically, investigation strategies are applied to ecotoxicological tests like those that evaluate the effects of environmentally reactive chemicals using single species of macroinvertebrates, zooplankton or algae in microcosm and mesocosm studies (reviewed in Fleeger et al., 2003; Connon et al., 2012). Further applications are the monitoring of toxic substances or organic pollutants in drinking water, or the determination of contamination, from heavy metals and other toxic chemicals, and the trophic status of agricultural or industrial waste water (reviewed in Bonada et al., 2006). For instance, freshwater bivalves (*Dreissena polymorpha*) or brown trout (*Salmo trutta*) from unpolluted sites were collected and exposed in polluted sites to assess the accumulation of dangerous substances (Camusso et al., 1994; Schmidt et al., 1999). The availability of sensitive indicator organisms is currently restricted to their applicability for ecotoxicological tests which mainly use algae and macroinvertebrates. These organisms are useful in detecting the lethal effects of environmental pollutants or water quality (Fleeger et al., 2003; Connon et al., 2012), but may be unsensitive to structural degradation, which is a crucial factor in European streams (Bernhardt and Palmer, 2011).

Structure-sensitive macroinvertebrate and algae species are currently not produced under standardised conditions in sufficient numbers for habitat assessments. In contrast to macroinvertebrates and algae, structure-sensitive fish species are intensively produced in aquaculture due to their high economic value for consumption (e.g. *Salmo salar*) or sport fishing (e.g. *S. trutta*). Consequently, all life stages (eggs, juveniles and adults) are easily available in high numbers and of standardised quality. Single life stages, such as eggs or larvae, are highly suitable to test the quality of specific habitats, e.g. river bed substratum or restoration effects of spawning grounds using active bioindication. Especially early life stages of salmonid fishes are sensitive to changes in water and substratum quality (Crisp, 1996; Rubin and Glimsäter, 1996; Soulsby et al., 2001; Jungwirth et al., 2003; Denic and Geist, 2010; Sternecker and Geist, 2010). However, numerous incubator systems have been developed to produce salmonids for conservation issues (reviewed by Kirkland, 2012), but it appears that only two of these systems (Pander et al., 2009; Pander and Geist, 2010b) have been specifically developed and used for bioindication to test restoration success. In addition to the availability of incubation systems adapted to bioindication requirements, the production of other potential bioindicators, such as freshwater mussels, is still a challenge.

5.2. Indicator systems for bioindication (active bioindication)

Like the bioindicator species themselves, indicator systems also need to fulfil several criteria. Materials and construction should facilitate sensitive detection of changes in environmental gradients

Table 3
Examples of commonly used bioindication assessment tools and indices in stream ecology.

Indicator	Assessment tool	Abbreviation	Literature	Region	MG	SG	RTR	Metrics included	Topic
Phytobenthos	Acification Index Periphyton	AIP	Schneider and Lindstrøm (2009)	Unlimed rivers, Norway		X	Species	Sensitivity to acidification	River acidification
Phytobenthos	Periphyton Index of Trophic Status	PIT	Schneider and Lindstrøm (2011)	Norway		X	Species	Sensitivity to eutrophication	Trophic status
Macrophytes and Phytobenthos	Macrophyte and phytobenthos based evaluation system for running waters	Phylyb	Schaumburg et al. (2006)	Germany	X		Species	Trophy, structural degradation, acidification, salinisation	Classification of the ecological status of rivers
Macrophytes	Trophic Index of Macrophytes	TIM	Schneider and Melzer (2003)	Germany		X	Species	Sensitivity to eutrophication	Water quality, trophic status
Macroinvertebrates	Saprobic Index	SI	e.g. for Germany Zelinka and Marvan (1961), Rolauffs et al. (2003), Meier et al. (2006)	Europe		X	Species	Saprobic status	Saprobic status of rivers
Macroinvertebrates	Biological Monitoring Working Party	BMWP	Armitage et al. (1983)	Worldwide		X	Family	Tolerance to organic pollution	Organic pollution
Macroinvertebrates	Ephemeroptera, Plecoptera, Trichoptera	%EPT	Lenat (1988)	Worldwide		X	Order	Sensitivity to water quality and structural degradation	Water quality
Macroinvertebrates	Species at Risk	SPEAR	Liess and Von der Ohe (2005)	Germany		X	Species	Sensitivity to organic pollutants and pesticides, generation time, migration ability, emergence time	Toxic pollution
Macroinvertebrates	Macroinvertebrate-based evaluation system for running waters	PERLODES	Meier et al. (2006)	Germany		X	Species	Saprobic status, habitat preference, taxonomic composition, diversity, acidification	Classification of the ecological status of rivers
Macroinvertebrates	Macroinvertebrate-based nutrient biotic index	NBI	Smith et al. (2007)	North America		X	Species	Sensitivity to nutrient enrichment	Measure of nutrient enrichment
Freshwater fish	Index of Biological Integrity	IBI	Karr (1981)	North America		X	Species	Species composition, rich-ness, tolerance, hybridisation, trophic measures, health condition, age structure, growth, recruitment	Classification of the ecological status of rivers
Freshwater fish	Fish-based evaluation system for running waters	FIBS	Dußling et al. (2004)	Germany		X	Species	Habitat preference, reproduction, trophy, age structure, migration, fish region, dominance	Classification of the ecological status of rivers
Freshwater fish	European Fish Index	EFI	Fame Consortium (2004)	Europe		X	Species	Trophic structure, repro-duction, habitat, migration, disturbance tolerance	Classification of the ecological status of rivers
Freshwater fish	Fish regions Index	FRI	Dußling et al. (2005)	Germany, Austria		X	Species	Natural probability of fish to occur in different river regions	Classification of the ecological status of rivers
Freshwater fish, Macroinvertebrates, Phytobenthos	Rapid Bioassessment Protocols	RBPs	Barbour et al. (1999)	USA	X		Species	Richness measures, composition measures, tolerance measures, trophic/habitat measures	Classification of the ecological status of rivers
All	Species richness	S	Arrhenius (1921)	Worldwide			Adaptable	Number of species	Diversity
All	Shannon Index	H	Shannon and Weaver (1949)	Worldwide			Adaptable	Number of species and individuals	Diversity
All	Evenness	J	Pielou (1966)	Worldwide			Adaptable	Distribution of individuals on species	Diversity

MG, assessment includes multiple taxonomic groups; SG, assessment is based on a single taxonomic group; RTR, required taxonomic resolution for species identification to calculate the index.

without impacting the development of the indicator organisms. Additionally, the system should allow the assessment of spatially explicit survival rates and conditions of the indicators. Furthermore, the gathering of information regarding physicochemical changes during the exposure time should be feasible, in order to detect which factors contributed most to the assessment status or performance of exposed organisms. For this reason, it is necessary to equip indicator exposure systems with measurement units that allow a variety of spatially resolved measurements, such as were implemented in the "egg sandwich" (Pander et al., 2009) and the Salmonid Egg Floating Box (SEFLOB, Pander and Geist, 2010b). Both systems can be used for bioindication of water and substratum quality using salmonid eggs. The "egg sandwich is a system for assessing stream substratum quality by linking measurements of depth-specific salmonid egg hatching success and physicochemical water variables from the same sites within the interstitial zone. This can be interesting, for instance, in the assessment of spawning grounds where salmonid eggs can naturally be buried in different depth layers with different oxygen concentrations (Soulsby et al., 2001). Linking survival rates with chemical and physical measurements like substratum texture can result in mechanistic understanding of ecological processes which all contribute to the effectiveness of restoration measures. The SEFLOB is a modified version of an up-flow incubation tray, commonly used in fish hatcheries, that is used as a bioindication tool for the exposure of fish eggs in free-flowing water. It was used in a pre-restoration assessment of potential re-introduction of the highly endangered Danube salmon into rivers which formerly served as spawning habitats. The SEFLOB was found to be more sensitive than chemical measurements for detecting deficiencies in water quality that limit egg survival (Pander and Geist, 2010b). This is an example of the suitability of active bioindication tools in testing water conditions suitable for the target species before implementation of restoration measures or the reintroduction of the target species. Studies of the "egg sandwich" in artificial and natural spawning grounds of *S. trutta* and *Thymallus thymallus* (Pander et al., 2009) testing restoration effectiveness suggest that this indicator system is an easy and cost-effective tool to evaluate the ecological functionality of the streambed. The system is applicable for the evaluation of all types of substratum restorations which are currently practiced, such as gravel introduction, raking, washing, and relocation. The system also has the potential to be adapted for assessments in various biogeographic or fish ecological regions using different indicator organisms such as other fish species (e.g. rainbow trout (*Oncorhynchus mykiss* in North America) or macroinvertebrates (e.g. water quality and substratum quality sensitive organisms such as thick shelled river mussel (*Unio crassus*)).

5.3. Combining active and passive bioindication

While restoration of water or substratum quality can easily be assessed with the available tools, active bioindication assessments for other measures remain a great challenge. Well-established indicators and exposure systems are widely lacking for the evaluation of the effects of measures such as the introduction of dead wood, macrophytes, and boulders, and the establishment of shallow habitats, riparian wood or fish bypass channels. Due to the ambiguous spatial effectiveness and the structural complexity of these measures, it is difficult to link caged-bioindicator organisms to restoration effects. Consequently, it can be advantageous to combine active and passive bioindication strategies. For instance, the mark-recapture method (Ihssen et al., 1981) can deliver insights into the functionality of stream restoration measures when applied with stocked or naturally occurring individuals that freely move between their preferred habitats (Denic and Geist, 2010; Pander et al., 2011). The use of organisms with high structural requirements, like freshwater crayfish (e.g. *Astacus astacus*, see Lundberg, 2004) or bullhead (*Cottus gobio*, see Kottelat and Freyhof, 2007), have great potential as detectors of structural improvements with this method. Additionally, the differences in habitat preference of stocked versus wild fish (e.g. *S. trutta* in Pander et al., 2011) can be used to establish links between the autecology of species and restoration success.

5.4. The importance of passive bioindication and its limits

Passive bioindication can be advantageous with increasing complex restoration, e.g. on ecosystem scale (Figs. 1 and 2), because several combined measures and their ecological interactions can be evaluated comprehensively at different levels of biological organisation (from microorganisms to fish). Passive bioindication is also applicable for detecting major anthropogenic impacts, e.g. the disruption of the river continuum (Ward and Stanford, 1983; Mueller et al., 2011), which can be essential in determining the primary impacts (e.g. weirs) and the most-limiting deficiencies (e.g. degradation of habitats and change in biodiversity) during the pre-restoration monitoring. Since the response of biological communities to changing environmental conditions can vary between taxonomic groups and rivers (Johnson et al., 2006; Paavola et al., 2006; Heino, 2010; Mueller et al., 2011), assessments at the ecosystem scale should include several trophic levels of animals and plants. As demonstrated in Mueller et al. (2011), the combination of several levels of biological organisation can reveal even small-scale effects resulting from weir-introduced heterogeneity in water depth, substratum composition and water velocity.

The utility of traditional bioassessment methods (Table 3) in strongly altered or restored habitats can be limited by inventories of highly restricted species (Noss et al., 1995; Pander and Geist, 2010a; Pander et al., 2011), high numbers of ubiquitous species (Kirchhofer, 1995; Pander and Geist, 2010a) and increasing numbers of neobiota (Pyšek and Richardson, 2010). Most well-established bioindication systems do not consider neobiota (e.g. FIBS Dußling et al., 2004; PERLODES Meier et al., 2006) and cannot be applied to assess pre- and post-restoration status if abundances of indicator taxa are very low. Because of this, the compilation of data to single indices, as practiced for WFD compliance, cannot be recommended for the evaluation of restoration success. Additionally, in restoration programs with transnational scales or worldwide application, an evaluation system is necessary which is not dependent on the species inventory of a specific region. This can be the case for the restoration of the river continuum, where measures concerning fish passage, removal of weirs, and bank reinforcement, or the construction of more natural river courses have to be implemented in different geographical regions (e.g. Danube wide) or scales (tributary to catchment).

5.5. Stream restoration: from biological to ecological indicators

The multivariate analysis of habitat characteristics, community composition, and functional traits can be used as a universally applicable tool for the detection of restoration-induced habitat changes (Clarke, 1993; Mueller et al., 2011; Pander et al., 2011; Sundermann et al., 2011). The relative comparison of habitat characteristics, community composition, and functional traits between restored and unrestored sites allows the quantification of the effects of different instream habitat restoration measures (Pander and Geist, 2010a; Sundermann et al., 2011), or the assessment of artificial flow courses as functional compensatory habitats (Pander et al., 2011). Using several replicates of biotic community structure per treatment (restoration measure, habitat type), instead of one single sampling stretch, increases the statistical robustness of the results. This approach was suitable to distinguish habitats, detect

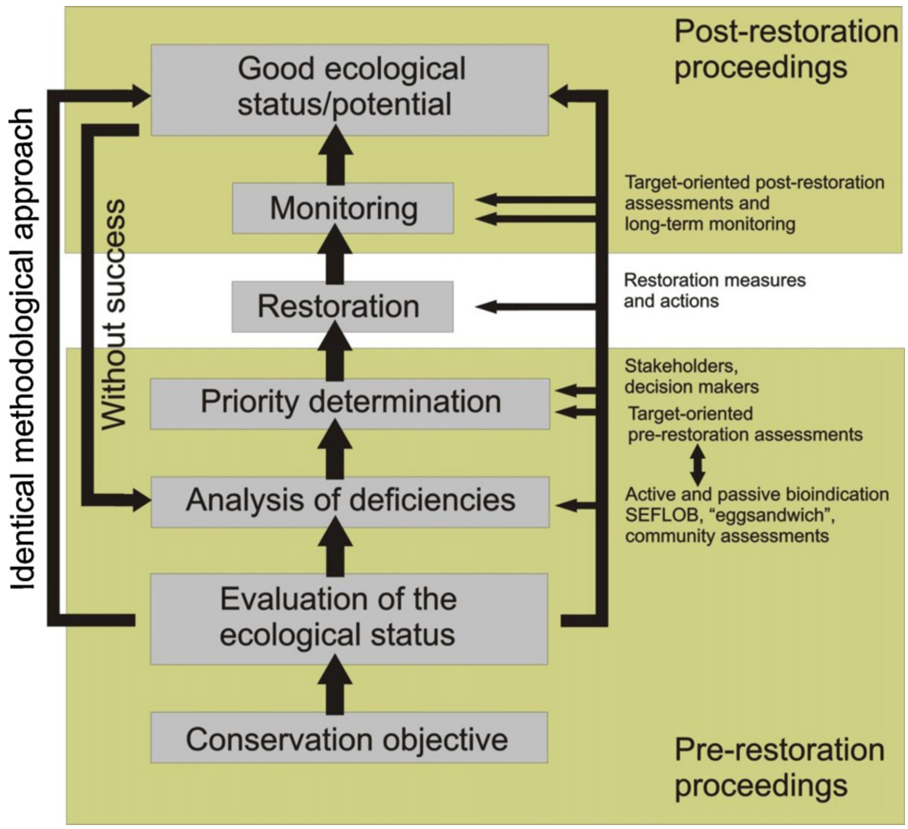

Fig. 2. The Proceeding Chain of Restoration (PCoR) is a step-by-step approach which systematically structures the complex procedure of restoration from pre-restoration proceedings, restoration measures and actions, to post restoration proceedings. Pre- and post-restoration proceedings are highlighted in green. Individual steps of the proceeding chain are highlighted in grey. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of the article.)

seasonal effects on colonisation of habitats, and to draw conclusions about restoration success and remaining ecosystem deficits. Furthermore, this data analysis strategy integrates biotic and abiotic data independent of the sampling method, variables assessed, river-specific species composition, and occurrence of indicator taxa or the target species. Consequently it can be applied across biocenotic regions.

Different data structures resulting from different sampling strategies and scales can be combined with multivariate analysis. For instance, sampling strategies that differ between taxonomic groups likely yield data with different structures (spatial resolution, number of species or individuals). Freshwater algae can be scraped off small areas of substrate while fishes have to be sampled with traps, net-fishing or electrofishing from larger areas. Additionally, data often differ in taxonomic resolution and degree of accuracy (presence absence data, relative abundances and quantitative data). The multivariate approaches like the multi-dimensional scaling (non-metric: NMDS, metric: MDS) which were tested in Pander and Geist (2010a) and Pander et al. (2011) can be applied for different data structures and combined analysis of several taxonomic groups (Mueller et al., 2011).

6. Recommendations for measuring restoration success: "The Proceeding Chain of Restoration"

Guidelines are urgently needed which include important criteria for ecologically successful restoration (Palmer et al., 2005) to improve river restoration success (Wohl et al., 2005). In most cases, river restoration does not follow a target-oriented procedure but is rather based on trial and error. Moreover, the linkage between restoration planning (restoration management), engineering (construction work) and ecology (ecological requirements) is poorly established. The Proceeding Chain of Restoration (PCoR) is meant to improve collaboration between project managers, restoration experts and scientists, by providing standardised guidelines that can be followed from the beginning of restoration planning to the evaluation of restoration success (Fig. 3). The PCoR is a step-by-step approach which systematically structures the complex procedure of river restoration and ensures scientific involvement early in the restoration process, as recommended by many authors (e.g. Wohl et al., 2005; Jansson et al., 2005). The approach is subdivided into three general parts, the pre-restoration proceedings, the implementation of measures, and the post-restoration proceedings.

6.1. The pre-restoration proceeding

As the first step in the pre-restoration proceeding, a clear objective has to be defined to determine the targets and scope of research used to set conservation priorities (Dahm et al., 1995; Jähnig et al., 2011). A clear conservation objective, specifically, becomes important when it is necessary to focus on one target species out of a pool of several species with high conservation value which compete for resources and space (Simberloff, 1998).

The evaluation of the pre-restoration status is considered a key stage in river restoration (Bernhardt and Palmer, 2011). It is the basis for the analysis of deficiencies and the determination of priorities, where scientists should ideally assist stakeholders to find the most effective measures (Wohl et al., 2005). The investigation of the pre-restoration status should generally take into account the conservation of target taxa as well as aquatic communities and the maintenance of river functions.

Since habitat conditions within small rivers can be very patchy and highly variable in space and time (LeRoy Poff and Ward, 1990; Winemiller et al., 2010; Braun et al., 2012), it is necessary

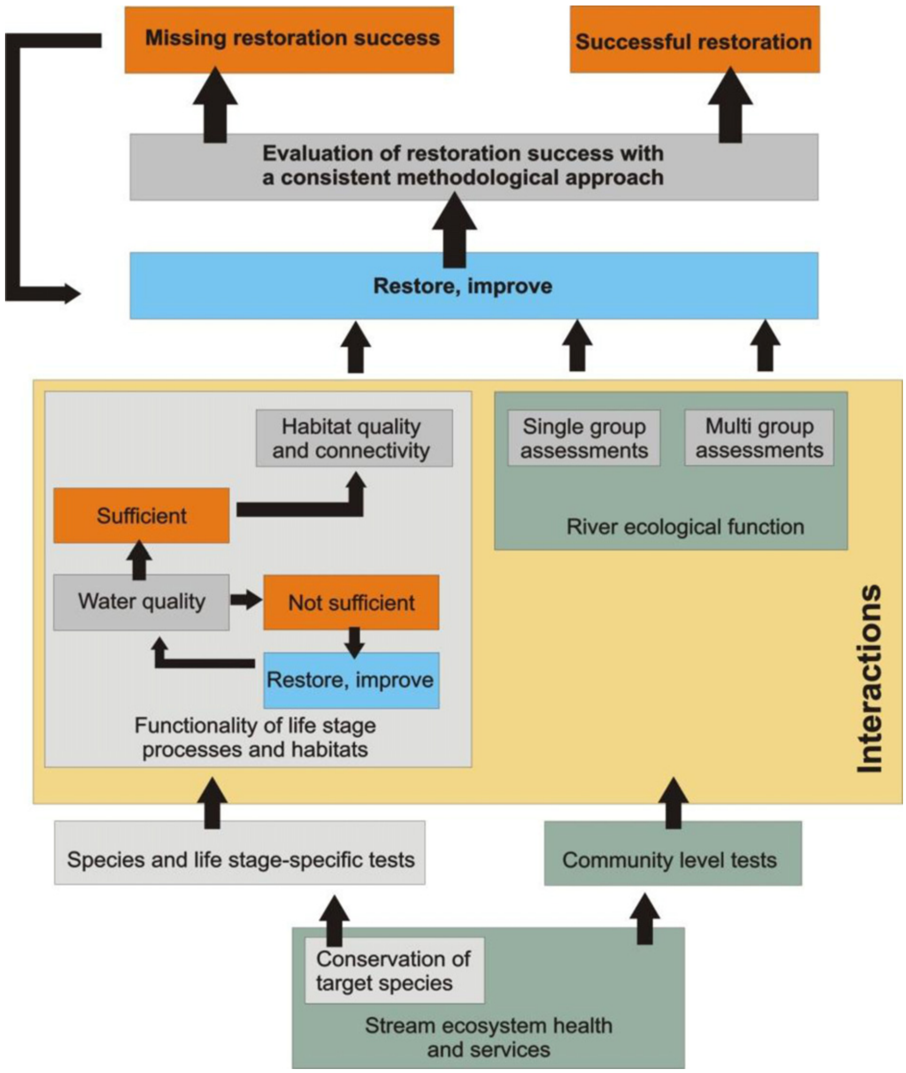

Fig. 3. Flow chart of an integrative efficiency control for restoration measures or ecosystem assessments. Black arrows indicate a step-by-step approach. The assessment strategy is subdivided in a component that focuses on target species (highlighted in grey) and a second component that focuses on ecosystem health and ecosystem services assessment (highlighted in green). The combination of assessment methods with different complexities enables the investigation of interactions between different levels of organisation (highlighted in yellow). The results of the assessment steps (highlighted in orange) can lead to different restoration measures and actions (highlighted in blue) or to previous steps. Note that restoration targets or all steps can address single target-species reference communities or a combination of those. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of the article.)

to evaluate the ecosystem status particularly before and after the restoration (Chapman, 1999). Considering initial (i.e. pre-restoration) conditions is increasingly important in all countries where potential reference sites are affected by land-use change and the only unaffected river stretches are in steeper upstream reaches. In this case, the pre-restoration stage can serve as a reference to move away from (Palmer et al., 2005). For example, if a weir-impounded river should be restored, the approach of Mueller et al. (2011) can be used to assess the pre-restoration status and to quantify subsequent changes.

In the stage in which deficiencies are evaluated, standardised methods and applications (Bernhardt et al., 2005) provide a framework for continuous control of implemented measures and actions. Analysis of deficiencies and evaluation of primary factors responsible for degradation leads consequently to the proposal of effective restoration measures and actions. Based on the results of the pre-restoration evaluations, the statistics applicable for data analyses then can be determined. For example, sampling design (e.g. number of samples, length of river stretch), degree of data transformation (e.g. square-root transformation, log-transformation), and statistical methods (e.g. non-metrtic multidimentional scaling, principle component analysis) should be specified in a standardised protocol for application in the post-restoration proceedings. In the stage of priority determination, the involved stakeholders and decision makers have to decide which restoration measures and actions will be most efficient to reach the conservation objective stated at the start of the project. This stage basically provides the last possibility to discuss options concerning restoration measures or conflicts in conservation objectives.

6.2. *The post-restoration proceeding*

Every restoration has the potential for evaluating in the post-restoration proceedings which measure contributed most to the conservation objective (Downs and Kondolf, 2002). Therefore, it is not necessarily important to evaluate every project in order to gain knowledge of the functionality of restoration measures. In some cases, a subset of pilot studies may be sufficient if the same measurements are considered in comparable rivers (Bernhardt et al., 2007; Hering et al., 2010). Ideally, the subsets cover the assessment of a broad range of restoration types, e.g. restoration of water quality (Pander and Geist, 2010b), bank habitat restoration (Pander and

Geist, 2010a), spawning ground restoration (Pander et al., 2009) and fish migration (Pander et al., 2011). To determine successful restoration strategies, assessment tools and applications for data analyses used in the pre-restoration stage, should be applied again following a consistent methodology. This can prevent data failures and misinterpretations and facilitate the segregation of restoration effects from other ecosystem processes. Knowledge of the main drivers leading to successful restoration will avoid trial-and-error proceedings and can lead to systematic target-oriented stream restoration.

Monitoring results should be published internationally, nationally and regionally to enable restoration experts to profit from the findings and to improve further restoration planning (Nienhuis and Gulati, 2002). Expert knowledge should be collected, transformed into a generally understandable form and presented on easily accessible platforms (e.g. in the case of Europe, web-sites of national conservation authorities, Climate-ADAPT climate-adapt.eea.europa.eu, BioFresh data.freshwaterbiodiversity.eu, WISE water.europa.de) to comply with legal requirements of public access to environmental information (Table 1) and for the involvement of a wider public audience.

7. New ways for an integrative assessment of target species-oriented restoration success and overall river ecological function

The restoration of river ecological function can address a broad variety of aspects at different scales e.g. riparian management (e.g. Hladyz et al., 2011b), floodplain reconnection (e.g. Opperman et al., 2009), water quality aspects (e.g. Osborne and Kovacic, 1993), bank stabilisation (e.g. Li and Eddleman, 2002), dam removal (e.g. Maloney et al., 2008), fish passability (e.g. Agostinho et al., 2002; Knaepkens et al., 2006), channel reconfiguration (e.g. Kondolf et al., 2007; Jähnig et al., 2010) and the reduction of fines from the catchment area (e.g. Roni et al., 2008). Additionally, different levels of biological organisation typically have divergent recovery rates and time (Power, 1999). For this reason, there is rarely one, ideal scale at which to conduct an ecosystem assessment that will suit several purposes (Millennium Ecosystem Assessment, 2005). Furthermore, the evaluation system needs to be adjustable to the conservation objective. Particularly, the assessment of overall ecosystem health (typically at larger spatial scales) requires a different approach than the assessment of life stage-specific restoration such as the improvement of spawning ground quality for salmonids or the mitigation of migration barriers (typically at small spatial scales). Consequently, a multiscale approach which uses large and small-scale assessments and the combination of several taxonomic groups simultaneously may deliver the most reliable and meaningful results. This includes active bioindication to test water quality and life stage-specific habitat quality in combination with passive bioindication-based community assessments to detect large-scale effects on ecosystem level (Fig. 3). Target species-based approaches can be useful to address restoration of major deficits in stream ecosystems, but need to consider all life stages and the interactions with other species (Geist, 2010). For instance, for the endangered salmonid *Hucho hucho* a population decline can be the result of a variety of deficiencies like insufficient water quality, limited habitat functionality, restrictions in spawning migrations, or reduced productivity due to disturbances in the food web (Geist et al., 2009). To point out the most limiting factors for salmonid conservation and the restoration process, a methodological approach should at first test the general suitability of water quality in the study stream (Pander and Geist, 2010b). Limiting water chemical conditions have to be eliminated before other restoration measures can be effective. After the improvement of e.g. the quality of industrial and domestic waste waters and the reduction of agricultural depositions from the catchment area in the study stream, the SEFLOB method (Pander and Geist, 2010b) should be applied again to test for potential improvements (Figs. 2 and 3). If water quality requirements are fulfilled then life stage-specific strategies and key-habitats should be considered next. This includes the assessment of spawning ground suitability (Pander et al., 2009), juvenile habitats, adult habitats and habitat connectivity (Pander and Geist, 2010a; Denic and Geist, 2010; Pander et al., 2011) which is important for spawning migrations and movements between life stage-specific habitats (Geist et al., 2009). As *H. hucho* is a top predator in stream ecosystems (Holčík, 1995) with naturally low species abundance, the quality of species-specific habitats may not be a universal indicator for the development of sustainable populations. Single- or multi-group assessments which include several trophic levels (Mueller et al., 2011; Pander et al., 2011) deliver important insights into other limiting factors like interactions between species, the integrity of food webs, or the productivity of stream sections (Douglas et al., 2005). Additionally, the integration of population genetic information for the target species, as suggested in the IFEBC-concept (Geist, 2011), can become particularly important for breeding programs, supportive stocking, and reintroduction. To choose the best population for reintroduction, active bioindication methods such as the SEFLOB and the “egg sandwich” can also be applied. With these systems, it is possible to test the performance of different strains and populations under distinctive environmental conditions of the study stream during their most critical life stages.

8. Conclusions

River restoration in industrial and developed countries is of high priority due to policy requirements. In most developing countries, there is still a strong need for the implementation of a legal framework similar to that outlined by European and North American legislation. A legal basis forces the evaluation of the ecological status of rivers in order to identify existing deficits in river function. This is essential for the development of national restoration concepts and the protection of important ecosystem services (drinking water, food supply, economy).

River restoration is currently often practiced as an unsystematic course of action. The restoration of stream ecosystem health and ecosystem services can be most successful when target-oriented, systematic, and integrative approaches are used to determine initial conditions and to measure restoration effects. A stepwise evaluation of the primary factors of disturbance or degradation may be most suitable when considering all major drivers of successful restoration.

The PCoR presented herein integrates the standardised approach for scientific monitoring into a step-by-step guideline for river restoration, politics, and management. Systematic approaches in stream restoration planning should follow the principle of comprehensiveness, adequacy, representativeness, and efficiency (CARE principle, Linke et al., 2011) to match the PCoR criteria. Target species (important as indicator, flagship, umbrella and keystone species) based approaches, as presented herein and in Geist (2010), in combination with assessments of ecosystem processes can fulfil the criteria of the CARE principle. Merging the interactions between abiotic and biotic factors at different levels of organisation qualitatively and quantitatively, as proposed in the IFEBC concept (Geist, 2011), with practical conservation and the PCoR concept is likely to increase the efficiency of aquatic restoration and targeted ecosystem services in the future. Ultimately any effort to make restoration more effective will be beneficial to preserve aquatic biodiversity.

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