TECHNISCHE UNIVERSITÄT MÜNCHEN

Lehrstuhl für Aquatische Systembiologie

Bioindication tools for measuring the success of stream restoration

Joachim Wolfgang Pander

Vollständiger Abdruck der von der Fakultät Wissenschaftszentrum Weihenstephan für Ernährung, Landnutzung und Umwelt der Technischen Universität München zur Erlangung des akademischen Grades eines

Doktors der Naturwissenschaften

genehmigten Dissertation.

Vorsitzender: Univ.-Prof. Dr. A. Melzer

Prüfer der Dissertation:

1. Univ.-Prof. Dr. J. P. Geist
2. apl. Prof. Dr. R. Gerstmeier

Die Dissertation wurde am 12.06.2012 bei der Technischen Universität München eingereicht und durch die Fakultät Wissenschaftszentrum Weihenstephan für Ernährung, Landnutzung und Umwelt am 30.07.2012 angenommen.

Bioindication tools for measuring the success of stream restoration

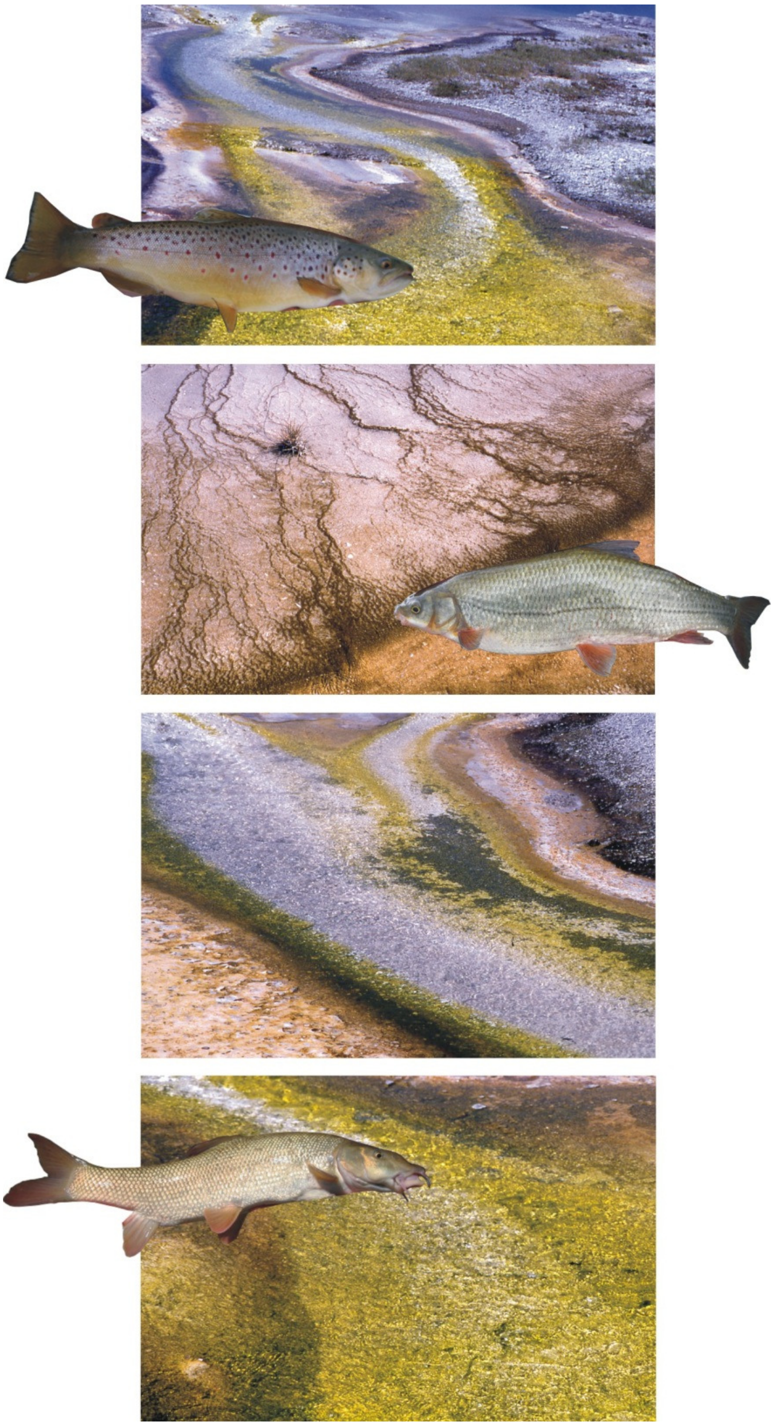


Table of contents

I List of Tables

II List of Figures

1 The need for river restoration and how to measure its success

3 Salmonid-egg floating boxes as bioindication for riverine water quality snd stocking success

4 The “egg sandwich”: a method for linking spatially resolved salmonid hatching rates with habitat variables in stream ecosystems

5 Seasonal and spatial bank habitat use by fish in highly altered rivers – a comparison of four different restoration measures

6 Ecological functions of fish bypass channels in streams: migration corridor and habitat for rheophilic species

7 The effects of weirs on structural stream habitat and biological communities

8 Synthesis: a new integrative approach to improve the proceeding and efficiency control of river restoration

III Preface

This PhD-thesis “Bioindication Tools to Measure the Success of Stream Restoration” is intended to contribute to a target oriented and more effective approach to measure stream restoration success and therefore help to protect aquatic biodiversity whilst contributing to an improvement of the ecological functionality of rivers.

The thesis is structured in nine chapters as follows: A general introduction (chapter 1) which describes the importance of aquatic biodiversity and the reaction of the European Legislation to its predicted loss is followed by sections which explain the complexity of river restoration and the role of bioindication in river restoration. In chapter 2 the main objectives of the work are stated. The following chapters (3-7) contain five case studies in which tools for active and passive bioindication were developed and described. In addition, for active bioindication two toolboxes are described: At first the assessment of water quality using a salmonid-egg floating box and afterwards the assessment of spawning ground quality through measures of spatial resolved salmonid egg hatching success. Methods for passive bioindication were developed through analysing restored bank habitats in a highly modified model stream, ecological functions of fish bypass channels as migration corridors, and habitats for fish and the effects of weirs on structural stream habitat and biological communities. In chapter 8 the five case studies are discussed in general and a new integrative and target oriented practice for river restoration and the assessment of river restoration success is presented. In this general discussion, challenges of using bioindication are linked to the synthesis of new ways for an integrative evaluation of target species oriented restoration success and overall river ecological function, also including a standardised restoration protocol called “The Proceeding Chain of Restoration” (PCoR).

Acknowledgements

This PhD-thesis could not have been realised without the knowledge and help of many people and I am grateful to all who contributed to the success of this work. Naturally the shortcomings of this thesis are mine and mine alone.

I am particularly grateful to my supervisor Prof. Dr. habil. Jürgen. Geist. Prof. Geist managed to support and guided me in a structured scientific course of action. Obviously, a long-term project like a PhD thesis has its temporarily motivational fluctuations: I am therefore very grateful for his ability to provide freedom in academic thinking while keeping up the necessary ambitional goals at the same time. His great help in providing access to and also limiting the number of questions which arose during the course of events leading to the final thesis has been of great importance and cannot be valued enough. His ideas and the countless discussions were of significant importance while developing the ideas mentioned in the following. Thank you very much. In addition, I also would like to include all the staff at the Aquatic Systems Biology Unit for the excellent working conditions, the atmosphere in an international research orientated group, and the numerous occasions I had in sharing and developing my ideas in discussions with my colleagues here. I particularly want to thank M. Ache, and J. Brandner who shared most of the time with me in the office. I thank Dr. B. Gum for reading earlier versions of the manuscript and M. Müller for her tireless efforts during almost all field sampling surveys, the on-going projects, and for her constructive contribution and the aid in providing the clear line of argumentation in areas where I almost got lost in the manuscript.

I would like to thank to the head of the PhD examination commission, Prof. Dr. A. Melzer and to the jury committee members Prof. Dr. habil. J. Geist, and Prof. Dr. R. Gerstmeier for their contributions. In addition my acknowledgements go to Dr. Dr. h.c. J. Bauer (Lehrstuhl für Tierhygiene) and Dr. habil. R. Kühn (AG Molekulare Zoologie) for their support in the early planning stages of the PhD-Thesis.

To carry out all the case studies many people contributed on a practical as well as theoretical stage to the success of this work. Extensive field sampling is often teamwork and so I want to thank everybody who contributed to the electrofishing surveys and the collection of physicochemical data. I thank J. Steinhilber for his support to organise the technical equipment for the electrofishing and to look after the laboratory references of the fisheggs. Since the field work required a variety of special licenses e.g. for the electrofishing and the accessibility of protected areas I am grateful to Dr. O. Born (Fachberatung für Fischerei, Schwaben), J. Hoch (Fachberatung für Fischerei, Niederbayern), Dr. W. Silkenat (Fachberatung für Fischerei, Unterfranken) and R. Lauer (Landkreis Hassberge). The validation of the eggboxes could not have been possible without the support of B. Hornauer (state fish hatchery Mauka) and P. Aschenbrenner (state fish hatchery Lindbergmühle) who provided the fisheggs for the field and laboratory surveys. I want to thank K.H. Billmeier and J. Schiener for their assistance during field survey for the "SEFLOB"-study and K. Sternecker for the provision of field data for the "egg sandwich". The habitat restoration in the river Günz was initiated by the local owner of the fishery rights, F.J. Schick and the investigation on the Nassach and the Riedbach was initiated by the head of the local fishery association, V.

German. I am grateful to Landesfischereiverband Bayern e.V. for financial support of these two case studies. In this context I want to thank Dr. U. Pulg and J. Schnell for the coordination of the funding. The introduction of the habitat restoration measures and the field work at the Günz would not have been possible without the commitment of R. Klocke and E. Eser from the ´Bayerische Elektrizitätswerke GmbH´. From all case studies the research on the effects of weirs on aquatic communities was the most intensive field work and I am grateful to following persons and institutions: I would like to thank ‘Fischereiverein Obere Saale Hof’, ‘Fischereiverein Miesbach-Tegernsee’ and J. Heinlein for their permission to carry out studies in their fishery rights and also would like to thank them for their local support.

Above all I want to thank my family for supporting the idea for my PhD-study and to look after Ferdinand whenever it was necessary. At last I would like to thank both the most important persons in my life: Jutta and Ferdinand. Jutta, I would like to thank you for your ongoing patience and understanding for the many days out of field sampling and the extra time I spent in the office when work had to be finished in time. You always listened appreciative to all the problems during this work, had an open ear to all the problems of a PhD candidate throughout the whole time and never stopped to support me. You never had a doubt that I would finally be able to successfully complete the work. I cannot thank you enough in words.

Finally I want to thank Ferdinand for his pure existence. Thank you for being here.

IV Summary

Freshwater resources are essential to sustain human existence and the alteration of rivers, lakes and wetlands has followed the economic development for centuries. As a consequence, freshwater biodiversity is critically threatened with stream ecosystems being most heavily affected, particularly in industrial countries. The European Government reacted to the predicted loss of biodiversity with the proclamation of the Water Framework Directive in the year 2000. To mitigate the deficits of all targeted rivers and streams in the proclaimed narrow time frame, it is essential to implement the most effective restoration measures. Consequently, it is necessary to know which restoration measures contribute most to the good ecological status or the good ecological potential.

The main objective of this study was to develop methodologies for the monitoring and evaluation of the success of stream restoration measures using active and passive bioindication. The key aspect was to develop toolboxes and applications for target species and life stage-focused assessments of water- and substratum quality and the evaluation of overall river ecological functioning and anthropogenic disturbance. A general focus was given to assessment strategies which improve the validation of restoration effects applying univariate and multivariate statistics. The findings of the presented case studies are integrated in a holistic approach to reach a target-oriented course of action for river restoration already in early planning stages, including the evaluation of river restoration measures. The studies include the development of two bioindication tools, in which salmonid egg hatching success and physicochemical water variables from adjacent sites can be used to determine riverine water and substratum quality. As first a salmonid egg floating box (SEFLOB) was tested for applicability in a pre-restoration assessment of water quality for the re-introduction of the highly endangered Danube salmon *Hucho hucho*. Secondly, the “egg sandwich”, an incubation system for the stream substratum was developed and tested in artificial and natural spawning grounds of two salmonid species of high conservation value (*Salmo trutta*, *Thymallus thymallus*). The results of the laboratory and field experiments suggest that both indicator systems are easy and cost-effective tools with a high reliable technical functionality to determine water and substratum quality of natural and artificial spawning grounds.

For the application of passive bioindication in ecological monitoring, standardised sampling designs were developed and applied in three case studies assessing the effects of stream habitat restoration, the conservation value of fish bypass channels and the effects of weirs on structural stream habitat and aquatic communities. In the first study, the fish community in a highly modified model stream ecosystem was used to test the applicability of relative comparisons of different restored bank habitats for the evaluation of restoration success. The investigation was standardised by assessing stream sections of equal length with the same methodology and a high number of replicates per restoration type. In the second study, the fish community of three nature-oriented bypass channels and their adjacent upstream and downstream sites was investigated to assess the role of fish passes as compensatory habitats and migration corridors for different fish species. The third study was designed to test the suitability of the taxonomic groups fishes, macroinvertebrates, macrophytes and algae as passive bioindicators for the quantification of the serial discontinuity introduced into streams by weirs in five different rivers. For the quantification, upstream and downstream sites of weirs were investigated with a sampling design comprising standardized length, number and arrangement of sampling points. The multivariate analysis applied in the passive bioindication studies turned out to be suitable to distinguish adjacent riverine habitat types, detect seasonal effects on colonization of habitats and to draw conclusions about restoration success and remaining deficits on ecosystem level. Due to the integration of biotic and abiotic effects and the independence of sampling methods, the investigated set of variables, the river specific community composition and the occurrence of target species, this data analysis strategy is transferable to environmental monitoring projects in general.

The results of the presented case studies suggest that a combination of target species based active bioindication and community based passive bioindication analysed with multivariate statistics seem to be most suitable for a holistic evaluation of restoration success including the monitoring of stream ecosystem health. Since the response of biological communities to changing environmental conditions can be inconsistent between taxonomic groups and rivers, assessments on ecosystem scale should include several levels of biological organisation. For a holistic and target-oriented course of action, a stepwise evaluation of the main impact factors of disturbance or degradation with increasing complexity from water quality assessments to the evaluation of river ecological function turned out to be most suitable to consider all major drivers of a successful restoration by simultaneously focusing stepwise on accuracy and extent of required information. These findings were combined to the proclaimed “proceeding chain of restoration” (PCoR) which considers all steps from the determination of the conservation objective to the post-restoration monitoring.

V Zusammenfassung

Die Verfügbarkeit von Süßwasser ist für den Menschen existentiell wichtig und die anthropogenen Veränderungen von Flüssen, Seen und Feuchtgebieten sind eng an die ökonomische Entwicklung der vergangenen Jahrhunderte gekoppelt. Als Konsequenz ist die aquatische Biodiversität heute stark bedroht. Besonders Fließgewässer in den Industrieländern sind sehr stark davon betroffen. Um dem vorhergesagten Verlust an aquatischer Biodiversität entgegen zu wirken, verabschiedete die Europäische Union im Jahre 2000 die Wasserrahmenrichtlinie. Das darin vorgegebene enge Zeitfenster zur Beseitigung der strukturellen und wasserchemischen Beeinträchtigungen macht es notwendig, die entsprechenden Renaturierungen so effektiv wie möglich durchzuführen. Dazu ist es unumgänglich zu wissen, welche Maßnahmen am meisten dazu beitragen den „guten ökologischen Zustand“ oder das „gute ökologische Potential“ der Gewässer zu erreichen.

Als übergeordnetes Ziel wurden in dieser Arbeit aktive und passive Bioindikationsmethoden für die Untersuchung und Bewertung von Fließgewässerrenaturierungen entwickelt. Im Einzelnen wurden Bioindkationssysteme erfunden und getestet, mit denen zielarten- und lebensstadienbasierte Untersuchungen der Wasser- und Substratqualität durchgeführt werden können. Weiterhin wurden Methoden ausgearbeitet um die ökologische Gesamtfunktionalität und die Auswirkungen von anthropogenen Veränderungen auf diese zu erforschen. Generell wurde dabei darauf geachtet, Auswertemethoden univariater und multivariater Statistik anzuwenden mit denen Effekte einheitlich quantifizierbar sind und sich besser statistisch absichern lassen. Aus den Ergebnissen der verschiedenen Fallstudien wurde eine ganzheitliche Handlungsanweisung abgeleitet, um eine zielgerichtete Fließgewässerrenaturierung von den ersten Planungsschritten an zu erleichtern.

Die ersten zwei Fallstudien umfassen die Etablierung von Inkubationsboxen zur aktiven Bioindikation der Wasser- und Substratqualität. In den Boxen wird der Entwicklungserfolg von Fischeiern in Verbindung mit wasserchemischen Messungen als Bewertungskriterium verwendet. Als erstes wurde eine Schwimmbox (SEFLOB) gebaut und im Vorfeld von Restaurierungsmaßnahmen eingesetzt, um die Wasserqualität für die Wiedereinbürgerung des stark in seinem Bestand zurückgegangenen Huchens (*Hucho hucho*) zu testen. Als nächster Schritt wurde das sogenannte “egg-sandwich“ zur Überprüfung der Substratqualität entwickelt. Bei diesem Inkubationssystem können die einzelnen Kammern für die Fischeier mit einer wasserchemischen Messeinheit gekoppelt werden, um die Substratqualität von natürlichen und künstlichen Kieslaichplätzen, unter Berücksichtigung eines Tiefengradienten, zu untersuchen. In dieser Studie wurden die Laichplätze von den zwei naturschutzfachlich bedeutenden Fischarten, Bachforelle (*Salmo trutta*) und Äsche (*Thymallus thymallus*), auf ihre Funktionalität hin überprüft. Zusätzlich wurde die Eibox unter Laborbedingungen getestet. Die Ergebnisse der Labor- und Feldversuche lassen den Schluss zu, dass beide Systeme zuverlässige und kosteneffiziente Werkzeuge mit hoher technischer Funktionalität zur Bewertung der Wasser- und Substratqualität von natürlichen und künstlichen Laichplätzen in Fließgewässern sind.

Zur Verbesserung und Erweiterung passiver Bioindikationsmethoden für ökologische Monitoringverfahren wurden drei weitere Fallstudien durchgeführt. Die Schwerpunkte lagen dabei auf Untersuchungen von Maßnahmen zur Restauration von Uferhabitaten in hochgradig veränderten Wasserkörpern und den Effekten von naturnahen Umgehungsgerinnen zur Wiederherstellung der Durchgängigkeit und ihrer Funktionalität als Ersatzhabitat. Als weiterer wichtiger Punkt wurde in einer der Fallstudien quantifiziert, wie groß die Auswirkungen von Wehren auf das Fließgewässerkontinuum und die aquatischen Lebensgemeinschaften sein können. In der Ersten Studie wurde in einem kanalartigen Modellgewässer anhand von relativen Vergleichen der Fischartenzusammensetzung aufgezeigt, welche ökologisch verbesserten Uferhabitate den größten Renaturierungseffekt aufwiesen. Dazu wurde der Gewässerrand in vier standardisierte Habitattypen eingeteilt, von denen jeweils gleich lange Abschnitte mit einer großen Zahl an Wiederholungen beprobt wurden. In der zweiten Studie wurde die Fischartenzusammenestzung von drei naturnahen Umgehungsgerinnen mit den zugehörigen Unterwasser- und Oberwasserstrecken der zu umgehenden Wehre erfasst und vergleichend in Bezug gesetzt. Damit konnte überprüft werden, wie wichtig diese Fließstrecken als Wanderkorridor für Fische sind und welchen Beitrag sie als Ersatzhabitat für verschiedene Fischarten leisten können. Das Untersuchungsdesign der dritten Fallstudie zur passiven Bioindikation wurde entwickelt, um die durch Wehre verursachte serielle Diskontinuität von Fließgewässern aufzuzeigen. Zur Quantifizierung der Effekte wurden die verschiedenen taxonomischen Gruppen Fische, Makroinvertebraten, Makrophyten und Periphyton mit standardisierter Länge, Anzahl und Anordnung der Messpunkte direkt unterhalb und oberhalb der Querbauwerke untersucht. Bei allen drei Studien zur passiven Bioindikation stellte sich heraus, dass die angewandten multivariaten Auswertemethoden sehr gut geeignet waren, unterschiedliche Restaurationseffekte aufzuzeigen. Dies erwies sich als besonders vorteilhaft, um saisonal bedingte Unterschiede der Habitatwahl, den Restaurationserfolg der Maßnahmen, sowie verbleibende Defizite auf ökosystemarer Ebene abzuleiten. Aufgrund der Möglichkeit biotische und abiotische Daten miteinander kombiniert auszuwerten, kann diese Vorgehensweise sehr gut auf andere Projekte übertragen werden. Die Unabhängigkeit der

Auswertemethoden von gewässerspezifischen Lebensgemeinschaften, dem Vorkommen von Zielarten des Naturschutzes, und den analysierten abiotischen Habitatfaktoren sind dabei vorteilhaft.

Als Synthese aus allen Fallstudien geht hervor, dass eine auf Zielarten basierende aktive Bioindikation in Kombination mit passiver Bioindikation sehr gut dazu geeignet ist, neben dem Erfolg von Renaturierungsmaßnahmen, auch ein umfassendes Gesamtbild der Lebensgemeinschaften in Fließgewässern zu erhalten. Da einzelne Organismengruppen in verschiedenen Fließgewässern unterschiedlich stark auf Störeinflüsse reagieren und ihre Erholungszeiten sehr variabel sein können, empfiehlt es sich für Untersuchungen auf ökosystemarer Ebene mehrere taxonomische Gruppen gleichermaßen zu Betrachten. Für eine ganzheitliche Bewertung der ökologischen Funktionalität von Restaurationsmaßnahmen und Fließgewässern im Allgemeinen, ist eine schrittweise Bewertung der hauptsächlichen Wirkfaktoren ausgehend von Untersuchungen zur Wasserqualität bis hin zu Untersuchungen zur ökologischen Funktionalität am besten geeignet. Gleichzeitig mit der zunehmenden Komplexität der Untersuchung können die dazu notwendigen Informationen portionsweise mit angemessener Inhaltstiefe generiert werden. Die gewonnenen Erkenntnisse wurden zu einer strukturierten zielgerichteten Vorgehensweise, dem PCoR - Prinzip (proceeding chain of restoration, PCoR) zusammengeführt. In diesem Handlungsvorschlag wurden alle wichtigen Schritte und ihre Inhalte, beginnend bei der Formulierung des Leitbildes bis hin zu einem erfolgreichen Monitoring einer durchgeführten Renaturierung, verknüpft.

1 The need for river restoration and how to measure its success

1.1 The importance of aquatic biodiversity

Freshwater ecosystems are hot spots for biodiversity (Strayer & Dudgeon 2010, Geist 2011) and are recognized to contain 6% to 10% of all species and one third of all vertebrate species worldwide (Dudgeon et al. 2006, Balian et al. 2008), yet they cover only 0.8% of the earth surface (Gleick 1996). Additionally to the richness in species diversity the high number of specialized endemic species (Revenga et al. 2005) is remarkable. Freshwater ecosystems are strongly linked to human settlements or agricultural or industrial land use (Convention on Biological Diversity, CBD, United Nations 1992), making them particularly prone to degradation. Freshwater resources are essential to sustain human existence and the alteration of rivers, lakes and wetlands has followed the economic development for centuries. This is most evident in central European countries with their typically high population density and early industrialization. As a consequence, freshwater biodiversity is critically threatened (Ricciardi & Rasmussen 1999, Jenkins 2003) with stream ecosystems being most heavily affected (Stein & Flack 1997, Pimm *et al.* 2001, Gleick 2003). For instance Nilsson et al. (2005) showed that the percentage of large river systems affected by dams is over proportionally high in Europe (88%) compared to the situation worldwide (77%, Dynesius & Nilsson 1994). More than 48% of freshwater mussel species, 22.8% of freshwater gastropods, 32.7% of crayfishes, 25.9% of amphibians and 21.3% of freshwater fishes in North America are threatened with a large number of taxa being at risk to disappear in the next century (Riccardi & Rasmussen 1999). Between 1970 and 2002, the biodiversity of freshwater fishes, macroinvertebrates and macrophytes declined about 55% (Naiman 2008). Two decades ago, the future extinction rates were already estimated to be five times higher than those of terrestrial animals and three times higher than those of marine mammals (Riccardi & Rasmussen 1999). Naiman (2008) stated, that the decline of biodiversity in terrestrial and marine ecosystems is much smaller (by a factor of 1.7) than in freshwater ecosystems. This means, that the loss of freshwater species runs as fast as the loss of biodiversity in tropical rainforests (1-8% loss per decade, Reid 1997) which are considered to be depleted faster than any other biome (Myers 1988). Freshwater biodiversity provides many ecosystem services like clean drinking water, nutrition or recreation values that have been closely linked to human well-being in the Millennium Ecosystem Assessment (2005).

1.2 The role of European legislation in the restoration of river ecological function

The threat of many aquatic organisms and the predicted loss of aquatic biodiversity found its way in the consciousness of policy makers and politicians in many countries. This is reflected in the solid fundament of legislation, rules and regulations to protect aquatic species and their habitats (Table 1.1). Some of these were only established in recent times.

The European government reacted to the predicted loss of biodiversity and human well-being with the proclamation of the Water Framework Directive (WFD) in the year 2000 (European Parliament 2000). The major target stated in this directive is to reach the “good ecological status” or the “good ecological potential” of all major surface waters (e.g. rivers with a catchment area of more than 10 km²) and in groundwater until the year 2015. All member states of the European Union were obliged to transpose the European directive into national legislation. This was realised in Germany with the new implementation of the Water Management Act in 2010 (Wasserhaushaltsgesetz, WHG). The revised form of the Water Management Act of 1957 (Bundesministerium der Justiz 2009, implemented as WHG 2010) now has a stronger focus on the ecological functionality of rivers and streams creating an urgent need of successful restoration of running waters within a narrow time frame. Besides the WFD, several European directives and national implementations (Table 1.1) related to the topic of river ecological function were implemented recently. For example, directives to regulate the treatment of water pollution by discharges of certain dangerous substances (WPD, European Parliament 2006) or the Floods Directive (FLD, European Parliament 2007) contain instructions regarding aspects of river health.

The purpose of the WFD is to enforce the implementation of a comprehensive and integrative approach to protect resources sustainably and to protect and develop the ecological functionality of all major surface waters. This exceeds the traditional approach of chemical water quality assessment and comprises the evaluation of overall ecosystem health with a strong focus on aquatic biodiversity. Consequently, the directive regulates an investigation tool to describe, assess and classify all directive-wide concerned water bodies and to enable the comparability of surface water conditions on a European scale. The classification contains three major types, natural surface water bodies (NSWB, rivers, lakes, transitional waters and coastal waters), heavily modified water bodies (HMWB, a natural river which is substantially changed in character as a result of physical alterations by human activity) and artificial water bodies (AWB, a body of surface water created by human activity).

Table 1.1: 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 and analogously the German implementation and the North American regulations.

[1] European Parliament (2000). Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy. *Official Journal of the European Union* **327**,1-73.
[2] Bundesministerium der Justiz (2009). Gesetz zur Ordnung des Wasserhaushalts (Wasserhaushaltsgesetz - WHG). *BGBl.* **I**, p. 2585.
[3] 92nd United States Congress (1972). Federal Water Pollution Control Amendments of 1972 (CWA / Clean Water Act). Public Law 92-500.
[4] European Parliament (1992). Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. *Official Journal of the European Union* **206**, 7-50.
[5] Bundesministerium der Justiz (2009). Gesetz über Naturschutz und Landschaftspflege (Bundesnaturschutzgesetz - BNatSchG). *BGBl.* **I**, p. 2542.
[6] 93rd United States Congress (1973). An Act to provide for the conservation of endangered and threatened species of fish, wildlife, and plants, and for other purposes (ESA). Public Law 93-205.
[7] European Parliament (1985). Council Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment. *Official Journal of the European Union* **175**, 40-48.
[8] Bundesministerium der Justiz (2010). Gesetz über die Umweltverträglichkeitsprüfung (UVPG). *BGBl.* **I** p. 94.
[9] Bundesministerium der Justiz (2011). Baugesetzbuch (BauGB). *BGBl.* **I**, p. 1509.
[10] 91st United States Congress (1970). National Environmental Policy Act of 1969 (NEPA). Public Law 91-190.
[11] European Parliament (1991). Council Directive 91/271/EEC concerning urban waste water treatment. *Official Journal of the European Union* **135**, 40-52.
[12] Bundesministerium der Justiz (2004). Verordnung über Anforderungen an das Einleiten von Abwasser in Gewässer (Abwasserverordnung- AbwV). *BGBl.* **I**, pp. 1108, 2625.
[13] 94th United States Congress (1976). Toxic Substances Control Act (TSCA). Public Law 94-969.
[14] European Parliament (2006). Directive 2006/11/EC of the European Parliament and of the Council on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community. *Official Journal of the European Union* **64**, 52-59.
[15] European Parliament (2008). Directive 2008/1/EC of the European Parliament and of the Council concerning integrated pollution prevention and control. *Official Journal of the European Union* **24**, 8-29.
[16] Bundesministerium der Justiz (2002). Gesetz zum Schutz vor schädlichen Umwelteinwirkungen durch Luftverunreinigungen,Geräusche, Erschütterungen und ähnliche Vorgänge (Bundes-Immissionsschutzgesetz - BImSchG). *BGBl.* **I**, p. 3830.
[17] Bundesministerium der Justiz (2011). Gesetz zur Förderung der Kreislaufwirtschaft und Sicherung der umweltverträglichen Beseitigung von Abfällen (KrW-/AbfG). *BGBl.* **I**, p. 1986.
[18] 80th United States Congress (1947). Federal Insecticide, Fungicide, and Rodenticide Act of 1947 to regulate the marketing of economic poisons and devices, and for other purposes (FIFRA). Public Law 80-104.
[19] European Parliament (1991). Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. *Official Journal of the European Union* **375**, 1-8.
[20] Bundesministerium der Justiz (2007). Verordnung über die Anwendung von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten und Pflanzenhilfsmitteln nach den Grundsätzen der guten fachlichen Praxis beim Düngen (Düngeverordnung - DüV). *BGBl.* **I**, p. 221.
[21] European Parliament (1998). Council Directive 98/83/EG on the quality of water intended for human consumption. *Official Journal of the European Union* **330**, 32-54.
[22] Bundesministerium der Justiz (2011). Verordnung über die Qualität von Wasser für den menschlichen Gebrauch (Trinkwasserverordnung -TrinkwV). *BGBl.* **I**, p. 2370.
[23] 93rd United States Congress (1974). An Act to amend the Public Health Service Act to assure that the public is provided with safe drinking water, and for other purposes (SDWA). Public Law 93-523,
[24] 107th United States Congress (2002). Public Health Security and Bioterrorism Preparedness and Response Act (Bioterrorism Act). Public Law 107-188.
[25] European Parliament (2007). Directive 2007/60/EC of the European Parliament and of the Council on the assessment and management of flood risks. *Official Journal of the European Union* **288**, 27-34.
[26] Bundesministerium der Justiz (2005). Hochwasserschutzgesetz (Gesetz zur Verbesserung des vorbeugenden Hochwasserschutzes). *BGBl.* **I**, p. 1224.
[27] 89th United States Congress (1965). Flood Control Act (FCA). Public Law 89-298.
[28] European Comission (2011). http://ec.europa.eu/environment/nature/conservation/species/redlist/
[29] Bundesamt für Naturschutz (2011). http://www.bfn.de/0322_rote_liste.html/
[30] 93rd United States Congress (1973). An Act to provide for the conservation of endangered and threatened species of fish, wildlife, and plants, and for other purposes (ESA). Public Law 93-205.
[31] International Union for Conservation of Nature and Natural Resources (2011) http://www.iucnredlist.org/
[32] European Parliament (2003). Directive 2003/4/EC of the European Parliament and of the Council on public access to environmental information and repealing Council Directive 90/313/EEC. *Official Journal of the European Union* **41**, 26-32.
[33] Bundesministerium der Justiz (2004). Umweltinformationsgesetz (UIG). *BGBl.* **I**, p. 3704.
[34] 99th United States Congress (1986). Emergency Planning and Community Right-to-Know Act (EPCRA). Public Law 99-499.

Table 1.1

European Directive	Code/Year	W	H	S	Content	National Regulations (examples): Germany	USA
Water Framework Directive[1]	WFD 2000/60/EC	X	X	X	Protection, restoration and long-term sustainable use of clean water	Wasserhaushaltsgesetz (WHG) 2010[2]	Clean Water Act (CWA) 1972[3]
The Habitats Directive (Natura 2000)[4]	FFH 92/43/EEC		X	X	Maintenance of biodiversity, taking account of economic, social, cultural and regional requirements	Bundesnaturschutzgesetz (BNatSchG) 2010[5]	Endangered Species Act (ESA) 1973[6]
Environmental Impact Assessment Directive[7]	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 Umweltverträglichkeitsprüfung (UVPG) 2010[8]; Baugesetzbuch (BauGB) 2011[9]	National Environmental Policy Act (NEPA) 1969[10]
Urban Waste Water Directive[11]	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) 2009[12]	Clean Water Act (CWA) 1972[3]
Water Pollution by Discharges of Certain Dangerous Substances Directive[14]	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[2]; AbwV 2004[12]	Toxic Substances Control Act (TSCA) 1976[13]
Integrated Pollution Prevention and Control Directive[15]	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) 2002[16]; Kreislaufwirtschafts- und Abfallgesetz (Krw-/AbfG) 2011[17]; WHG 2010[2]; AbwV 2009[12]	Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) 1947[18]; Toxic Substances Control Act (TSCA) 1976[13]; Clean Water Act (CWA) 1972[3]
Nitrates Directive[19]	NID 91/676/EEC	X			Pollution prevention of nitrates from agricultural sources concerning ground and surface waters	Düngeverordnung (DüV) 2007[20]	No nitrate specific regulation besides CWA[3]
Drinking Water Directive[21]	DWD 98/83/EC	X			Consumer health protection and ensurance of wholesome and clean water	Trinkwasserverordnung (TrinkwV) 2011[22]	Safe Drinking Water Act (SDWA) 1974[23]; Public Health Security and Bioterrorism Preparedness and Response Act (Bioterrorism Act) 2002[24]
Floods Directive[25]	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 vorbeugenden Hochwasserschutzes (Hochwasserschutzgesetz) 2005[26]	Flood Control Act (FCA) 1965[27]
European Red List[28]	ERL IUCN 2010 ERL			X	Detailed and up to date information on biodiversity and conservation status of species	Rote Liste gefährdeter Tiere Deutschlands[29]; BNatSchG 2010[5]	Endangered Species Act (ESA) 1973[30]; IUCN Red List of North American Threatened Species[31]
Public Access to Environmental Information[32]	PAII 2003/4/EC				Establishment of a general right of any person to environmental information held by public authorities	Umweltinformationsgesetz (UIG) 2004[33]	Emergency Preparedness and Community Right-to-know Act (EPCRA) 1986[34]

The "Good surface water status" means the status achieved by a surface water body when both, its ecological status and its chemical status are at least "good". The chemical status is evaluated by investigating chemical quality components (concentrations of pollutants defined by environmental quality standards established in the WFD, European Parliament 2000 Annex IX). The ecological status or potential is determined by investigating four biological quality components: fishes, macroinvertebrates, macrophytes, phytobenthos/phytoplankton as well as supporting physicochemical and hydromorhological quality components (WFD, European Parliament 2000 Annex V).

The European countries were commissioned to survey the status of surface waters and in a following step to develop River Basin Management Plans until 2009. River basin management plans are water policy plans based on catchment areas and drainage systems, which constitute 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 the WFD goals. The progress on implementation of programs of measures has to be reported in the year 2012. For the report of the proceeding success of improving the ecological status, it has to be evaluated in which way the already implemented restoration measures were successful or not.

1.3 The complexity of river restoration success

River restoration has a long tradition with numerous actions undertaken lately (Kondolf et al. 2007). Over the last 30 years, river restoration has become a widely applied approach to restore freshwater ecosystems (Bernhardt et al. 2007) and will 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 was estimated to exceed one billion dollars per year (Bernhardt et al. 2005) underlining the economic importance. Regardless of the immense financial input and the numerous restoration projects, surprisingly little is known about the drivers which determine successful restoration (Palmer et al. 2005). Since river restoration can be influenced by ecological, technical and socio-economic factors which all interact in a complex way (Figure 1.1) it may be difficult to find the “silver bullet” solution to determine restoration success. Many factors affect the restoration practice, but their impact is strongly correlated with the complexity of the restoration target. With increasing spatial scale and ecological complexity of the restoration project, different factors can become crucial for the implementation of measures. For instance, the restoration of complex ecosystem processes is strongly linked to extended recovery time (Power 1999). In developed countries, an increasing spatial scale of the restoration can lead to rising numbers of restrictions (e.g. agricultural land use, infrastructure, disposal systems and utility services) which can limit the feasibility of the project. These restrictions can cause an enhanced requirement of skills, an intensified stakeholder involvement and an increasing monitoring effort. All these factors contribute to rising project costs. Consequently, restoration on 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 on population level can be short (Power 1999). Additionally an easier technical implementation of the measure and high feasibility go along with comparatively small costs. Limited funding can affect the monitoring of the restoration measures much more than the implementation of the restoration itself (Minns et al. 1996), causing insufficient investigated success rates of river restoration. The lack of standardized methods for cost-effective monitoring of ecosystems or the assessment of restoration measures may exacerbate the problem.

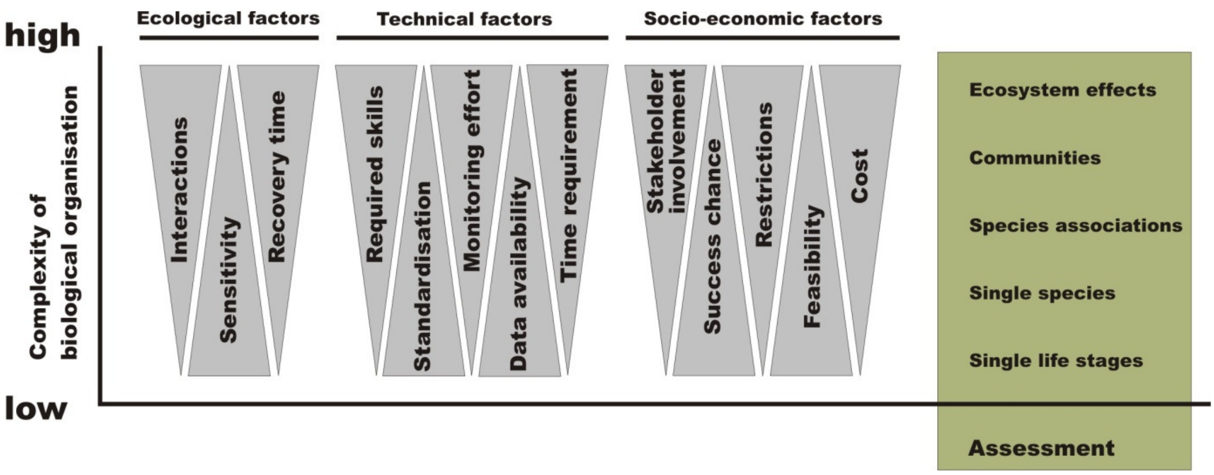


Fig. 1.1: River restoration is affected by ecological, technical and socio-economic factors which are displayed as grey triangles. The impact of the factor is increasing or decreasing depending on the complexity of biological organization of the restoration. The respective assessment scales for restoration targets of increasing biological complexity are highlighted in green.

The high complexity of river restoration and the different intentions for the implementation of restoration measures like the improvement of local recreation, flood protection or the ecological status, makes it difficult to focus on a universal term (Palmer et al. 2005, Bernhardt et al. 2007). Consequently, the success of restoration is difficult to measure. For each intention, measurement scales of success are variable with sometimes contrary definitions. In particular, the success for recreation or flood protection is not necessarily correlated with the improvement of river ecological function. While better recreation possibilities can be measured by enhanced visitor numbers, the success of flood protection by the reduction of flood damages is the improvement of ecological integrity a multidimensional construct of complex processes and interactions which all contribute to river ecological function. In this study the term “restoration success” refers to the criteria proposed in Palmer et al. (2005) were five major points with emphasis on ecological perspectives were stated. Besides the measureable improvement of flow dynamic processes, improved ecological condition and river health these criteria also address self-sustainability as well as resilience to external perturbations. Furthermore it is important that the implemented measures produce no lasting harm (first stated by Leopold (1948)) and that pre-and post-restoration assessments were carried out and the results were published.

Irrespective of terminological uncertainties and the complexity of restoration projects an intensive monitoring could enrich the knowledge about the drivers of successful restoration. In the last two decades, the Bavarian water authorities alone spent more than 300 Millions of Euros (Table 1.2) for the implementation of river restoration projects. The intentions for these projects varied from flood protection, compliance with national legislation, improvement of recreational values, to the restoration of the ecological functionality of rivers. Besides the driver ecology, flood protection and legislation were frequently named reasons for the restoration. Under the term “legislation” all projects were classified which refer to the WFD-goals or attendant legislation. In principle, the WFD framework ensures and regulates the evaluation of the ecological status and the functionality of these restoration measures. However, so far there has been little or no outcome which factors contributed most to a successful ecological restoration. A web survey of the official web-pages from Bavarian water authorities revealed that for 86% of the projects, no data are available to determine failure or success of the implemented measures due to the lack of any efficiency controls (Table 1.2). 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) evaluated in Table 1.2 were monitored on a short-term basis. None of these projects were monitored for more than one year. Large scale measures (more than 1 km restored river or bank length) seem to be monitored more intensively than small scale measures. The percentage of investigated large scale measures is almost double the size of small scale measures. Furthermore for only 7% of the considered large scale restorations a long-term monitoring was carried out. However, with a proportion of 25% large scale projects monitored in general, these efforts are still insufficient. In only 4% of all restorations the monitoring program included investigations of the pre-restoration status which is an important proxy to evaluate the project success holistically (Palmer et al. 2005).

Without a systematic and target-oriented evaluation of the restoration measures, it is not feasible to detect the highest impact factors driving restoration success. Consequently, it will be difficult to identify what the most powerful restoration strategy will be in future projects.

Freshwater conservation planning should therefore follow the CARE principle (comprehensiveness, adequacy, representativeness and efficiency), stated by Linke et al. (2011). This is increasingly important for the following proceedings of the WFD when the evaluation of the initial status of the good ecological conditions or potentials of rivers and lakes are already completed and the implementation of restoration measures is in process.

Table 1.2: Assessment of river restoration measures in Bavaria 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 effort were also drawn from the web. Restoration measures were classified in 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 where 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 passability	52 [51%]	11 [39%]	41 [56%]
	Structural improvements	65 [64%]	25 [89%]	40 [55%]
	Overall ecosystem health	22 [22%]	16 [57%]	6 [8%]
Drivers	Floodprotection & 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] [11.000-146.000.000]	12.204.524 [n = 21] [153.000-146.000.000]	113.068 [n = 29] [11.000-1.800.000]

To follow the WFD- directives, the restoration efforts have to be tested for their effectiveness to monitor the further status. Generally the strategy to use biological indicators for the monitoring of surface waters is suitable to detect environmental conditions (Bellinger & Sigee 2010). But up to now it is unclear if the WFD protocol is suitable for the detection of the success of restoration measures which can be highly variable in type and scale.

1.4 The role of bioindication in river restoration

Biological indicators can be a particular species or group of species whose function, population, or status can be used to determine ecosystem or environmental changes (Dziock et al. 2006). Depending on the indicator organism, its reaction can differ in sensitivity, ranging from changes in physiology, behaviour or morphology to death. On the basis of these reactions it is possible to draw conclusions about the ecological integrity of an ecosystem or the influence of potential dangerous substances (Dziock et al. 2006). In contrast to chemical and physical measurements, an advantage of bioindication is that animals and plants have to cope with changing or fluctuating environmental conditions during exposure and consequently integrate all environmental factors. While chemical and physical measurements to monitor ecosystem changes can be very cost intensive due to the high numbers of variables to be measured and the acceleration of measurements with increasing length of the survey period, bioindication can provide easy and cost-effective tools to acquire short-term and long-term information on the ecological integrity of environments and ecosystems (Neumann et al. 2003). The characteristics of a good indicator depend on the objectives of the environmental issue being addressed. However suitable and effective indicators must fulfill general criteria like economic and logistic suitability and biological efficiency (McGeoch 1998). This also includes a narrow ecological range, rapid response to environmental conditions, a well-defined taxonomy and reliable identification, a wide range in their geographic distribution and low costs (Bellinger & Sigee 2010).

Bioindication can be classified in many different ways: Some authors proclaim the classification due to the main application of bioindicators, e.g. state of the organisms which were classified, the main topic which is in focus of the investigation, the functional traits or the matrices which were used for the assessment (Knoben et al. 1995, Dziock et al. 2006). A more general and easy way to understand bioindication is the approach to distinguish between passive and active bioindication. Passive bioindication includes the assessment of present species or species communities occurring in an environment or ecosystem, whether it is natural or not (Schubert 1991). A classical application of passive bioindication is the saprobic index, where naturally occurring macroinvertbrate species are used to determine water quality (Zelinka & Marvan 1961). Active bioindication uses particular species which are exposed in natural environments, or under laboratory conditions (Schubert 1991, Dziock et al. 2006). It offers the possibility to standardize methodology with respect to exposure duration, size, age classes or the general quality of indicator organisms (Knoben et al. 1995). This investigation strategy is typically applied on ecotoxicological questions like the evaluation of effects of environmental reactive chemicals using macroinvertebrates, zooplankton and algae in single species tests, microcosm and mesocosm studies (Fleeger et al. 2003). Further applications are the monitoring of drinking water for toxic substances or organic pollutants, or to control agricultural or industrial waste water for its trophic status or contamination with heavy metals and other toxic chemicals. For instance, freshwater bivalves (*Dreissena polymorpha*) or brown trout *(Salmo trutta)* from unpolluted sites were collected and exposed in polluted sites for a particular time to assess the accumulation of dangerous substances (Camusso et al. 1994, Schmidt et al. 1999).

The usability of freshwater organisms for bioindication has been known since the middle of the 19th century (Bellinger & Sigee 2010). Today fish, invertebrates, macrophytes, algae and parasites are commonly used to monitor the status of freshwater ecosystems and many indicator systems for the condition of aquatic systems and for the evaluation of human impacts on them have been developed (Table 1.3). The monitoring program of the WFD uses passive bioindication for the biotic quality elements fishes, macroinvertebrates, macrophytes, phytobenthos/phytoplankton. Only in some cases the WFD-evaluation system as well as the other traditional established indices (Table 1.3) can be used to determine the success of small-scale restoration. None of them is practicable to detect improvements of target species-oriented restoration, where life stage-specific requirements and distinctive abiotic habitat variables have to be matched to determine success. The assessment of overall ecosystem health (typically on larger spatial scale) requires a different approach than the assessment of life stage-specific restoration like the improvement of spawning ground quality for salmonids, or the mitigation of migration barriers (typically on spatial small scales). For example, the restoration of spawning sites for rheophilic fish species like *Salmo trutta* and *Hucho hucho*, whose population status is of great value for reaching the WFD goals are often on a small spatial scale. The success of spawning ground functionality cannot be measured with the standard assessment tool for fishes for the ecological status of the WFD. This tool records the presence or absence of species and age classes in a relatively large spatial resolution which cannot be attributed to the improved functionality of spawning grounds and the successful development of these target species specific life stages. Therefore it is essential to explore how active and passive bioindication can be used to detect small scale, species and life stage-specific effects as well as large scale effects like the interruption of the river continuum (Ward & Stanford 1983) or the suitability of artificial flow courses as compensatory habitats for riverine biodiversity.

Table 1.3: Examples of commonly used bioindication assessment tools and indices in stream ecology. 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.

Indicator	Assessment Tool	Abbreviation	Literature	Region	MG	SG	RTR	Metrics included	Topic
Phytobenthos	Acification Index Periphyton	AIP	Schneider & Lindstrøm 2009	Unlimed rivers, Norway		X	Species	Sensitivity to acidification	River acidification
Phytobenthos	Periphyton Index of Trophic Status	PIT	Schneider & 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 & Melzer 2003	Germany		X	Species	Sensitivity to eutrophication	Water quality, trophic status
Macroinvertebrates	Saprobic Index	SI	e.g. for Germany Zelinka & 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 & 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
Freshwater fish	Index of Biological Integrity	IBI	Karr 1981	USA		X	Species	Species composition, richness, 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, reproduction, 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 & Weaver 1949	Worldwide			Adaptable	Number of species and individuals	Diversity
All	Evenness	J	Pielou 1966	Worldwide			Adaptable	Distribution of individuals on species	Diversity

2 Objectives

The main objective of this study was to develop methodologies for the monitoring and evaluation of the success of stream restoration measures. The key aspect was to develop easy and cost-effective toolboxes for a target species and life stage focused active bioindication assessment of stream restoration measures for fishes. Additionally, standardised methodologies using passive bioindication were developed for the monitoring of restoration measures, overall river ecological function and anthropogenic disturbance. A general focus was given to assessment strategies which improve the validation of the results by univariate and multivariate statistics. The five case studies concerning fundamental topics of measuring stream restoration success were integrated in a holistic approach to reach a target-oriented course of action for river restoration already in early planning stages, including the evaluation of river restoration measures.

In particular, the developed methodologies for active bioindication were focused on the evaluation of water quality and the success of spawning ground restoration for species of high conservation value such as *Salmo trutta* and *Hucho hucho*. This includes two bioindication tools, in which salmonid egg hatching success and physicochemical water variables from adjacent sites were used to determine riverine water and substratum quality. Both tools were tested for their applicability in field experiments and laboratory setups.

For the application of passive bioindication in ecological monitoring, standardized sampling designs were developed and applied in studies assessing the effects of stream habitat restoration, the conservation value of fish bypass channels and the effects of weirs. The fish community in a highly modified model stream ecosystem was used to test the applicability of relative comparisons of different restored bank habitats for the evaluation of restoration success. The investigation was standardised by assessing stream sections of equal length with the same methodology and a high number of replicates per restoration type. To assess the role of fish passes as compensation habitats for fishes, the fish community of three nature-oriented bypass channels was compared with river stretches of the same length in their respective upstream and downstream sites of the weirs. The functionality of fish passes as migration corridors for different fish species was assessed using standardized fish-trap surveys. The suitability of the taxonomic groups fishes, macroinvertebrates, macrophytes and algae for the quantification of the serial discontinuity introduced into streams by weirs was tested in five different rivers. For the quantification, upstream and downstream sites of the weirs were investigated with a sampling design comprising standardized length, number and arrangement of sampling points.

3 Salmonid-egg floating boxes as bioindication for riverine water quality and stocking success

J. Pander & J. Geist
Journal of Fish Biology 2010, **76**, 2584–2590

3.1 Abstract
The salmonid-egg floating box provides an easy bioindication tool for an assessment of water quality, as demonstrated here for the reintroduction of Europe's largest salmonid species, the huchen *Hucho hucho*.

3.2 Introduction
Water quality of North American and European rivers and streams has greatly improved over recent decades, resulting in an increase in available habitat for fish species that are sensitive to water pollution. This improvement offers a great potential for the conservation and reintroduction of endemic species into their historical ranges. In particular, the status of salmonids, such as the huchen *Hucho hucho* L., has been closely linked to water quality and may benefit from these improvements. Restoration of freshwater habitats has received both attention and funding over recent decades (Malakoff 2004). In the U.S.A. and Europe, huge financial resources have been spent to restore, protect or support declining salmonid stocks (Lichatowich 1999, Wu & Skelton-Groth 2002). There is still no consensus on the factors determining successful and sustainable habitat and fish stock restoration (Giller 2005, Palmer et al. 2005), and many reintroduction experiments are carried out on a trial-and-error basis, which is partly caused by a lack of suitable and cost-effective measures for a pre-assessment of reintroduction success. For sustainable and cost-effective salmonid reintroduction projects, the use of a bioindication test system is highly recommended for an assessment of the suitability of water quality for the most sensitive life stages before carrying out stocking activities. The endangered target species of this study, *H. hucho*, is one of the world's largest salmonid species (Holčík 1990). It is endemic to the Danube River drainage. *Hucho hucho* distribution within the Danube River drainage is currently restricted, and conservation measures for reintroduction of the species, which integrate genetic and ecological data, have been proposed (Geist et al. 2009). This article describes a bioindication tool for riverine water quality using salmonid egg hatching success and physicochemical water variables from adjacent sites within the free-flowing water. The applicability of this toolbox was successfully tested in a laboratory exposure set-up (used as a reference) and for

the reintroduction of *H. hucho* into three calciferous and siliceous rivers in southern Germany.

3.3 Material and Methods

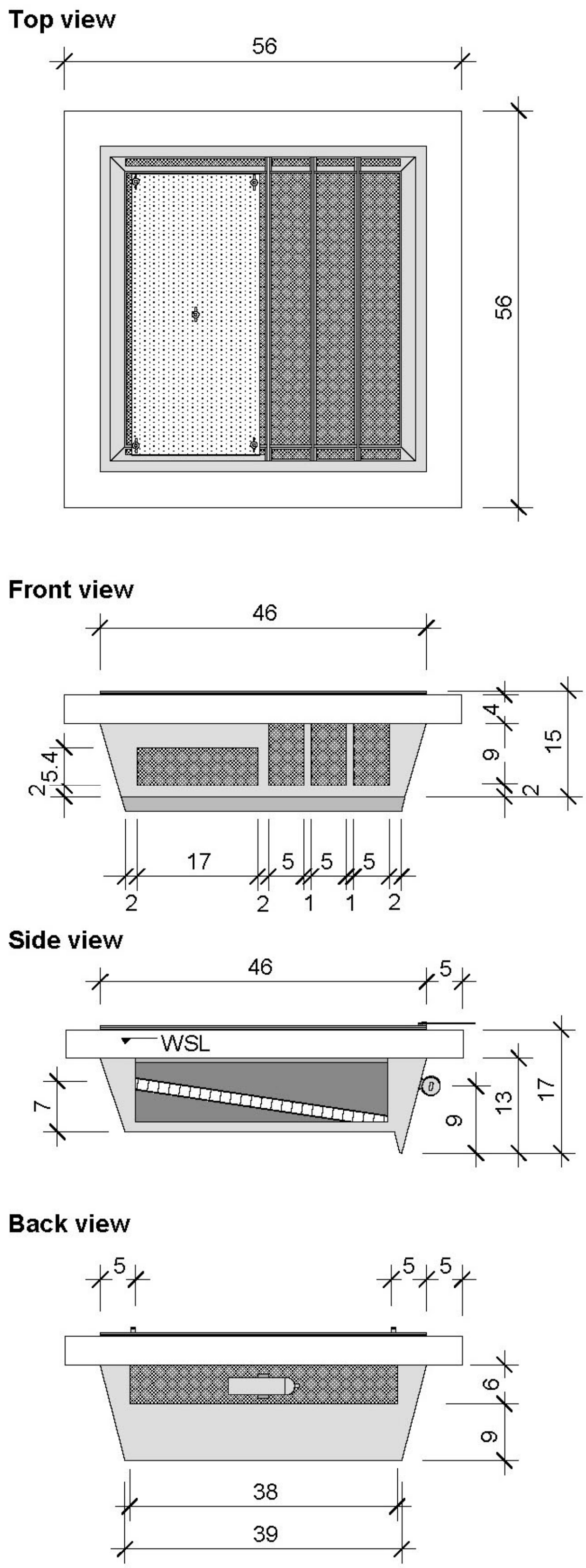


Fig. 3.1: Construction schematic with dimensions (cm) of the salmonid-egg floating box (SEFLOB), (WSL, water surface level). Top view without hardcover to show bulk exposure and grid inlay, floor space 6 x 39 cm, floor space grid partitioning 19 x 39 cm, front view with water inlets for grid partitioning and bulk exposure, side view is opened to show correct position of the grid inlay and back view with outlet and data logger.

A simple bioindication system for an integrative assessment of water quality by testing the hatching success of *H. hucho* was developed for this study. The construction of the salmonid-egg floating box (SEFLOB) was based on modified plastic upflow incubation trays, which are commonly used in salmonid hatcheries (Fig. 3.1). The SEFLOB was subdivided into two sections of the same size. One section was evenly divided into three replicates with 100 eggs in each. To check the effect of single exposed eggs *v.* the effects of the simultaneous bulk exposure of 100 eggs per replicate, the same number of eggs was also inserted into a grid in the second half of the box, where individual eggs were separated as previously described for the ‘egg sandwich’, a device for testing stream substratum properties (Pander et al. 2009). The front of the box was cut out and closed again with perforated aluminium plates. To keep the box floating and to ensure a constant water level inside the box, a circumferential floating body of 40 mm thickness, made from high-resistant foam, was attached under the upper frame with construction glue (Dichten & Kleben, www.obi.de). The box was closed with a plastic hardcover and locked with two cable clips. The separating plates and the lid were made of 3 mm plastic plates. Three swim boxes per site were anchored with a metal rod and safety lines. For long-term measurement of water variables [water temperature loggers from Lascar Electronics Ltd (www.lascarelectronics.com) were used], a data recorder was placed on the back of each box using a plastic clip (Fig. 3.1, back view).

The SEFLOB was tested in a laboratory set-up and by field tests in April and May 2009, including three sites in natural rivers within the original distribution range of *H. hucho*: the Moosach (MO, 48° 23´ 39.22´´ N; 11° 43´ 26.65´´ E), the Sickergraben (SI, 48° 35´ 06.56´´ N; 12° 13´ 28.76´´ E) and the Mitternacher Ohe (MH, 48° 50´ 23.91´´ N; 13° 18´ 52.38´´ E). The laboratory reference (HA) was exposed to groundwater at the Unit of Functional Aquatic Ecology and Fish Biology in which constant hatching rates >90% had been observed for brown trout *Salmo trutta* L. in previous years. At each of the test sites, three swim boxes were placed at a distance of 3 m from each other. The selection of the three test sites was based on geological differences (calciferous: MO and SI, siliceous: MH) as well as on their differences in their temperature regime due to different altitudes (Table 3.1). The *H. hucho* eggs were collected and fertilized directly in the state fish hatchery Lindbergmühle (Lindberg, Zwiesel, Germany). This facility maintains a parentage broodstock of known genetic constitution (Geist et al. 2009), which was provided for this study. A random mix of eggs from the same batch of spawners (two females and three males) was used for all 12 boxes. Two hours after fertilization, 100 eggs for each replicate were counted and placed into grid units and the transportation buckets. Eggs were then loaded into the SEFLOB directly at the study sites, where the boxes had been placed the previous day. To avoid any bias due to different transportation times, two teams were operating simultaneously and time between fertilization and exposure in the stream was 3 h for MH and 5 h for HA, MO and SI. Boxes were inspected and cleaned from floating debris on a daily basis. Dead eggs from the bulk exposure were removed daily. During egg exposure, water samples were taken from the boxes with a mobile 100 ml syringe at four stages: loading of the box, eye-point stage, hatching and the end of the exposures when the yolk sack of the larvae was consumed. The sampling was similar to the procedure of sampling interstitial water in stream substratum as described in Geist & Auerswald (2007). Water samples were analysed for pH, conductivity, temperature, dissolved oxygen, nitrite, nitrate and ammonium. The variables pH, conductivity (SC), temperature (T) and dissolved oxygen (O_2) were directly measured in the field using hand-held WTW-meters (WTW, www.wtw.com). Water samples were analysed for nitrite

(NO_2^-), nitrate (NO_3^-) and ammonium (NH_4^+) (Testkit Merck, www.merckchemicals.de) in the laboratory of the Unit of Functional Aquatic Ecology and FishBiology within 2 h. Hatching rates (HR) were calculated as per cent fish hatched alive out of the original number of eggs exposed. Normality of data was tested with the Kolmogorov–Smirnov test and Shapiro-Wilk's test and the homogeneity of variances was tested with Levene test. Since data and residuals were not always normally distributed, Kruskal–Wallis ANOVA and, in case of significance, Wilcoxon signed ranks *post hoc* tests were carried out to test for differences in mean values between the four assessed rivers. To test for significant differences between exposure of eggs in grids and bulk egg exposures, ANOVA was performed using a pooled data set over all study sites. All statistical analyses were carried out using SPSS 11 (SPSS Inc., www.spss.com), except the analyses of water temperature which were carried out using the freeware programme R (www.r-projekt.org). Significance was accepted at $p < 0.05$.

3.4 Results

The day sum degrees to the hatching time averaged 241 days° C and ranged between 197 (HA) and 281 days° C (MH); they were written in the range of values described for *H. hucho* by other authors (e.g. 280 days° C: Harsányi (1982); 224–240 days° C: Holčík (1995) and 260–300 days° C: Bohl (1999)). These differences may be explained by the higher temporal resolution of temperature measurements in this study (sampling intervals of 5 min, resulting in n = 8069–9500 measurement time points per stream) compared with previous studies. The lowest water temperatures were in MH and hatching in this stream occurred 4 days later compared with the other rivers, where hatching took place within ±1 day (Table 3.1).

Hucho hucho HR were consistent among replicates, but considerable variation occurred in the different study streams (Fig. 3.2). Highest HR were observed from eggs exposed to groundwater in the laboratory reference (treatment HA, mean ± s.d. HR = 79.6 ± 6.6%). A similar HR occurred in the River Moosach (treatment MO, mean ± s.d. HR = 78.8 ± 10·2%). Significantly lower hatching rates (Wilcoxon, $p < 0.05$) were found in the Mitternacher Ohe (treatment MH, mean ± s.d. HR = 23.9 ± 12.6%) and Sickergraben (SI, mean ± s.d. HR = 70.9 ± 11.5%). Water chemistry revealed significant differences between treatments for O_2, T, SC and NO_3^- (Table 3.1). The lower flow velocity in the laboratory set-up compared with the stream exposures did not seem to influence hatching success. The low HR in MH may be explained by water chemical differences between this stream and the other rivers. MH had lowest SC, highest mean NO_2^- values and lowest T, resulting in prolongation of the egg exposure time until hatching. High loads of pollen (farina) contributed to clogging of the inlets and outlets of the SEFLOB and formed a thin layer on top of the exposed eggs in MH, which may be an additional explanation for the low hatching success. Generally, analyses of water variables tend to be limited by their temporal resolution, since peak concentrations affecting survival in between samplings can easily be missed and since the effects of interaction of various factors on egg survival are still poorly understood. In contrast, the direct measurement of egg survival to hatch is a more integrative bioindication method, which can provide first valuable information on water quality suitability for the target species.

The comparison of overall mean HR of the grid inlays (mean $\pm$ s.d. 56.0 $\pm$ 34.4%) and the directly exposed eggs (mean $\pm$ s.d. 66.1 $\pm$ 22.3%) showed no significant differences (ANOVA, $p > 0.05$) and indicate that no systematic correction factor must be applied when using grid inlay exposure instead of direct egg exposure (Fig. 2). This finding was additionally supported by the absence of significant differences (Wilcoxon, $p > 0.05$) between grid exposure and free-egg exposure when conducting pair-wise comparisons among individual boxes and sites.

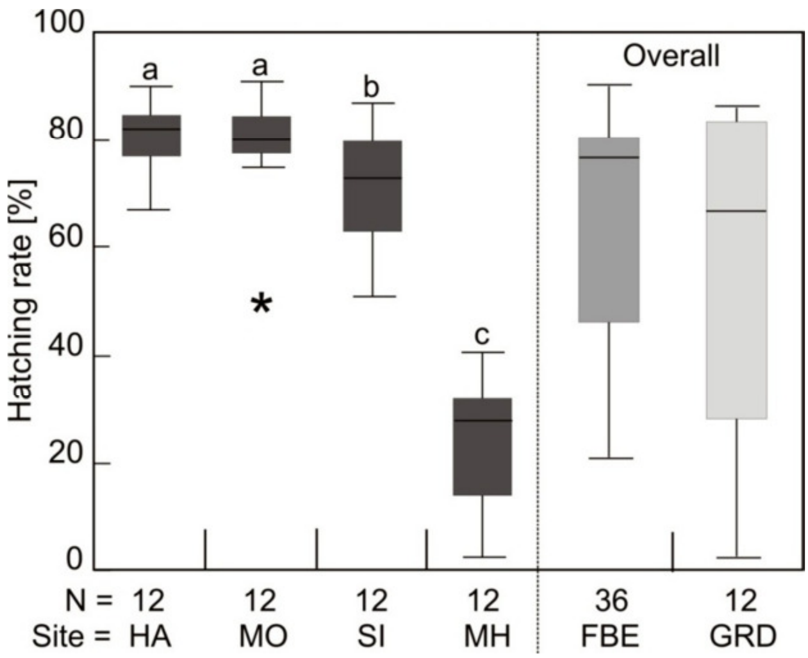


Fig. 3.2: Comparison of hatching rates of *H. hucho* in the four different treatments and between the bulk exposure and grid inlay: hatchery (HA), Moosach (MO), Sickergraben (SI), Mitternacher Ohe MH, free bulk exposure (FBE) and grid inlay exposure (GRD), N = number of replicates (comprising four compartments per box). Different lowercase letters (a, b, c) indicate significant differences ($p < 0.05$); box plots show the median and the interquartile range, * represents outlier (> 1.5 fold interquartile range).

Table 3.1: Characteristics of the four different study sites hatchery (HA), Moosach (MO), Sickergraben (SI) and Mitternacher Ohe (MH): mean flow velocity (V) measured 10 cm in front of the box (upper line) and mean flow velocity 10 cm behind the box (lower line), dissolved oxygen content (O_2), water temperature (mean values from the data logger) (T), pH, conductivity (SC), nitrite (NO_2^-), nitrate (NO_3^-), ammonium (NH_4^+) and degree days to hatching of *Hucho hucho* DC°. Different lowercase superscript letters (a, b, c) indicate significant differences ($p < 0.05$).

	V [ms^{-1}]	O_2 [$mg\ l^{-1}$]	T [° C]	pH	SC [µS cm^{-1}]	NO_2^-	NO_3^-	NH_4^+	DC°
Site	mean [range]	mean [range]	mean [range]	mean [range]	mean [range]	mean [range]	mean [range]	mean [range]	mean/range
HA	0.01^{a} [0.00 - 0.02] 0.01^{a} [0.00 - 0.02]	8.8^{a} [8.5 - 9.0]	11.7^{a} [11.5 - 12.0]	7.1 [6.9 - 7.2]	1002^{a} [983 - 1014]	0.07 [0.02 - 0.21]	13.9^{a} [5.7 - 20.7]	0.13 [0.03 - 0.41]	239 [197 - 280]
MO	0.21^{b} [0.07 - 0.51] 0.08^{b} [0.02 - 0.28]	11.1^{b} [10.1 - 11.6]	11.7^{b} [8.0 - 15.5]	7.6 [7.0 - 7.9]	746^{b} [671 - 772]	0.17 [0.06 - 0.54]	24.8^{b} [11.3 - 29.5]	0.10 [0.01 - 0.18]	239 [210 - 268]
SI	0.27^{b} [0.10 - 0.69] 0.08^{b} [0.02 - 0.13]	10.0 [8.8 - 11.1]	11.7^{c} [8.5 - 17.0]	7.3 [7.1 - 7.6]	602^{c} [586 - 616]	0.13 [0.06 - 0.24]	17.4^{a} [12.1 - 20.4]	0.15 [0.06 - 0.28]	237 [204 - 269]
MH	0.17^{b} [0.04 - 0.52] 0.05 [0.02 - 0.12]	10.4 [9.1 - 11.2]	10.0^{d} [5.5 - 15.5]	7.1 [6.8 - 7.8]	137^{d} [129 - 151]	0.22 [0.08 - 0.32]	9.5^{c} [4.2 - 14.9]	0.10 [0.04 - 0.20]	249 [218 - 281]

3.5 Discussion

The results of this study show that the egg bioindication system presented here may be a powerful and cost-effective tool for an integrative pre-assessment of water quality suitability for sensitive life stages of *H. hucho* and other fish species before carrying out stocking or reintroduction programmes. Water quality as a summation variable is one of the most important factors in restoration of salmonid stocks (Rubin & Glimsäter 1996) and should thus be tested as a first step in conservation and reintroduction programmes. The use of bioindicator target species for such an assessment integrates all environmental variables at the same time and thus has a high predictive power. If low hatching rates are observed as evident for the MH in the present study, the underlying factors must be mitigated before carrying out any other measures. It has to be noted, however, that some circumstances that can result in reduced egg survival in the SEFLOB boxes without substratum might not affect natural spawning success (e.g. pollen load in MH), while other problems such as high fine sediment loads and low oxygen concentrations within the hyporheic zone of the target stream might be overlooked. Quality of the free-flowing water can differ significantly from conditions in the interstitial zone (Geist & Auerswald 2007), and hatching rates estimated by SEFLOB boxes in the free-flowing water are likely to be higher than those measured in the interstitial zone (Pander et al. 2009). Successful restoration of salmonid stocks often also requires the assessment and restoration of stream substratum conditions (Soulsby et al. 2001a, Dumas & Marty 2006, Pander et al. 2009) and structural habitat quality (Denic & Geist 2010). As hatching success is only one of many perquisites of successful reintroduction, species and life stage-specific tests can provide information on the suitability of different ecotones.

The system used here may also be suitable for the assessment of water quality for larval stages of salmonids or as bioindication tool using other species such as macroinvertebrates.

4 The ‘egg sandwich’: a method for linking spatially resolved salmonid hatching rates with habitat variables in stream ecosystems

J. Pander, J. Schnell, K. Sternecker & J. Geist
Journal of Fish Biology 2009, **74**, 683–690

4.1 Abstract

This paper describes the development of the ‘egg sandwich’, 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.

4.2 Introduction

Biodiversity in freshwater ecosystems is critically threatened globally (Ricciardi & Rasmussen 1999, Jenkins 2003) with stream ecosystems being most heavily affected (Stein & Flack 1997, Pimm et al. 2001, Gleick 2003). Increasing evidence suggests that the properties of the stream substratum have a strong effect on the overall health of stream ecosystems (Palmer et al. 1997, Geist & Auerswald 2007). Conservation efforts in salmonid habitats have traditionally focused on stream substratum and spawning site restoration (Grost et al. 1991, Acornley & Sear 1999, Milan et al. 2000, Soulsby et al. 2001b). In light of the strong interest in restoration and assessment of stream substrata quality and salmonid spawning grounds, there is a need to provide tools for integratively assessing physicochemical and biological indicators. Here, a method for assessing stream substratum quality by measuring depthspecific salmonid egg hatching success, and physicochemical water variables from adjacent sites within the interstitial zone is presented. The applicability of this ‘egg sandwich’ (ES) was successfully tested in the laboratory and in natural and artificially constructed spawning sites of brown trout *Salmo trutta* L. and grayling *Thymallus thymallus* L.

4.3 Material and Methods

The ES is composed of two principal subunits: an egg exposure unit and a unit for extracting interstitial water samples from the same substratum depth layers in which the eggs are exposed (Fig. 4.1). The egg exposure unit consists of an aluminium grid and two perforated aluminium plates on the outside, creating10 x 13 dice-like chambers. Each chamber has a volume of 3.375 cm^3, providing sufficient space for the hatched fry. In the test, one fertilized egg per chamber was exposed, resulting in 10–13 replicates distributed over different depth horizons, and a total of 112 exposed eggs per box. Five chambers are penetrated by stainless steel socket-head screws for fixing both units and thus cannot house eggs. The upper horizontal row serves as a visual indicator for monitoring the exposure depth of the box. A second unit for extracting interstitial water is attached to the egg exposure unit. Its construction resembles that of the egg exposure unit, but the grid is penetrated by three perforated PVC tubes for sampling interstitial water at pre-defined depth horizons. One end of the tubes is equipped with sliding sockets to which flexible hoses with a length of 1.2 m are attached. The other end is closed with an elastic joint seal. Hoses can be sealed and individually marked with different colour codes to ensure correct depth assignments of water samples extracted through the hoses. In practical tests, sampling at 20, 70 and 115 mm proved successful, but this system can be easily adapted for sampling in even more horizons or in different depths. ES size, the size of the chambers and the number of eggs exposed in each chamber can be varied according to the research question addressed.

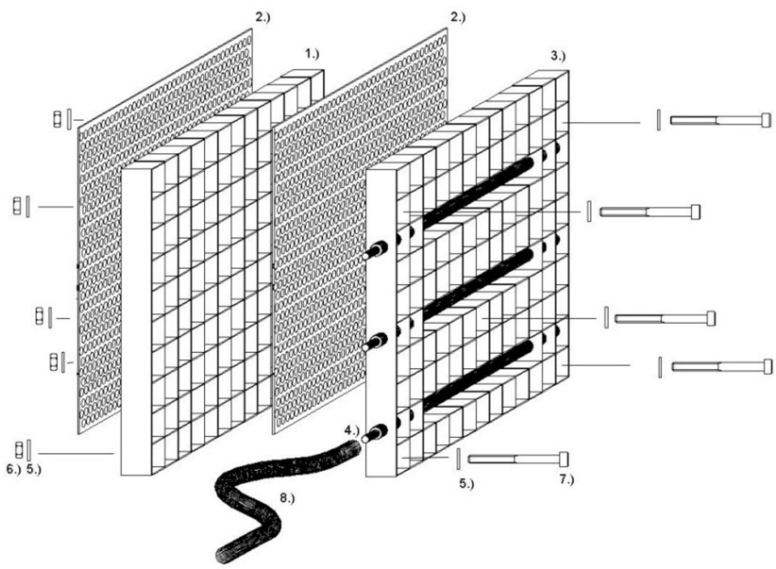


Fig. 4.1: Construction scheme of the “egg sandwich”; 1 = Aluminium grid (naturally anodised, length (L): 195 mm, width (W): 150 mm, depth (D): 15 mm, material thickness (MT): 0.5 mm, 130 chambers L: 15 mm, W: 15 mm, D: 15 mm), 2 = Perforated aluminium plate (L: 198 mm, W: 140 mm, MT: 1 mm, perforation bore diameter: 2 mm, partition: 3.5 mm diagonally lined), 3 = Aluminium grid (like 1, bore diameter for PVC-tubes 8 mm), 4 = perforated PVC-tube (L: 220 mm, inner diameter (ID): 5.5 mm, outer diameter (OD) 7.5 mm, one end with sliding socket OD: 5 mm, ID: 3.5 mm, other end sealed), 5= Flat washer (stainless steel, M4, OD: 19.5 mm, MT: 1.2 mm), 6 = Socket head screw (stainless steel, M4, L: 45 mm, length of thread: 20 mm, thread lead: 1.5), 7 = Hexagonal- or wing nut (stainless steel, M4, thread lead: 1.5), 8 = PVC-flexible hose (OD: 8 mm, MT: 1.5 mm, L: 1200 mm).

The ES is loaded with fertilized eggs by placing it into a shallow water tank. Ideally, the water level in the tank barely covers the grid. Individual chambers can be filled with fertilized eggs using a large core pipette or a turkey baster. After closing the lid, boxes should permanently stay immersed in cool, oxygenated water to prevent damage to the eggs. For the assessment of substratum conditions in a typical field setting, the egg-filled ‘sandwich box’ is vertically inserted into the stream substratum of the study site. For comparisons of conditions between free-flowing water and different substratum depths, it is recommended that the box is buried at a depth, which ensures that the upper box surface layer stays exposed to the free-flowing water conditions above the stream bed level. To ensure minimal disturbance of the native stream bed characteristics, a spade is used to create a small gap within the substratum into which the ES can be inserted (Fig. 4.2). Reference exposure of eggs in the free-flowing water (e.g. within a swimming box) allows for determination of specific development stages and hatching dates, which vary depending on the temperature day degree sum at the study sites.

During egg exposure, water samples are extracted by attaching a mobile 100 ml syringe to the hoses and by creating a vacuum, similarly to the procedure of sampling interstitial water in the stream substratum as described in Geist & Auerswald (2007). In a first step, the water volume entrapped inside the hose needs to be sampled and discarded before interstitial water from the defined substratum depths can be collected. The water volume has to be calculated or measured by the length and inner diameter of the hose. In the present exposures, water samples were analysed for pH, electric conductivity, temperature, dissolved oxygen, nitrite, nitrate, ammonium and redox potential. After hatching, the ES can be excavated and re-opened. Hatching success can be assessed according to the following criteria: (a) living fry, indicating favourable substratum conditions, (b) dead fry, indicating favourable substratum conditions during egg development but unfavourable conditions in the final exposure stage, (c) dead egg, indicating non-fertilized eggs or unfavourable conditions during early development and (d) missing egg due to predation, decomposition or erroneous loading of the chamber. An example evaluation is shown in Fig. 4.3. Further variables, such as siltation or clogging of chambers, can also be assessed at this stage, e.g. by taking photographs of the chambers and using computer-based image processing applications.

Applicability of the ES was tested by field and laboratory tests, including the following aspects: (1) mean hatching rates (HR) were compared between ‘egg sandwich’ exposure and reference egg exposure in regular upflow incubation trays (as typically used in salmonid hatcheries) under otherwise identical conditions (2) HR in the ES exposure were compared to the most commonly used field egg-exposure system, the modified Whitlock-Vibert boxes (WV-box; Whitlock, 1979; Mackenzie & Moring, 1988) at the same sites in the River Moosach (38°23´39´´ N; 11°43´26´´ E), (3) hatching rates of (1) and (2) were compared to

ES substratum exposed in a laboratory flume. Detailed descriptions of numbers of replicates are provided in Fig. 4.4. One-way ANOVA and Tukey post hoc tests with SPSS 11 (SPSS Inc., Chicago, IL, U.S.A.) were used to compare treatments. The spatial resolution of the ES exposure was resolved by testing pair-wise differences in HR between three different depth layers of stream substratum (20, 70 and 115 mm) exposed ES boxes in the River Moosach. All investigations were carried out in winter 2007 to 2008.

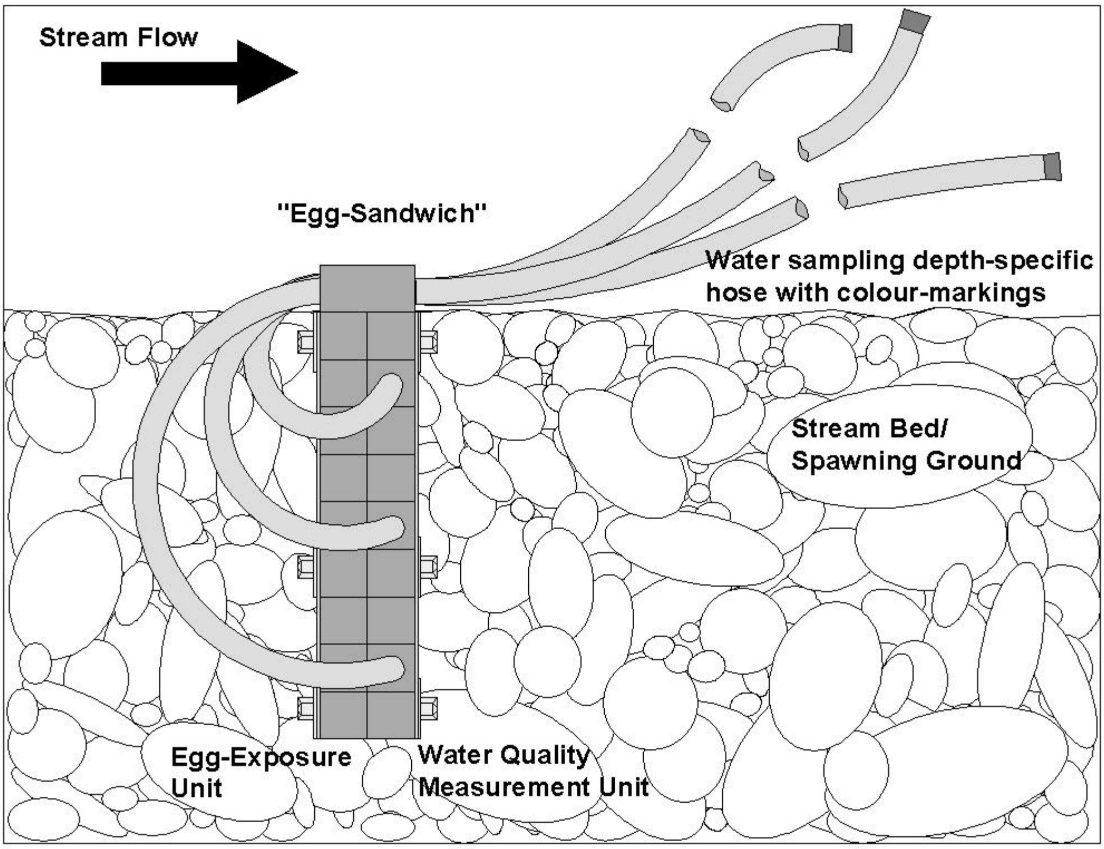
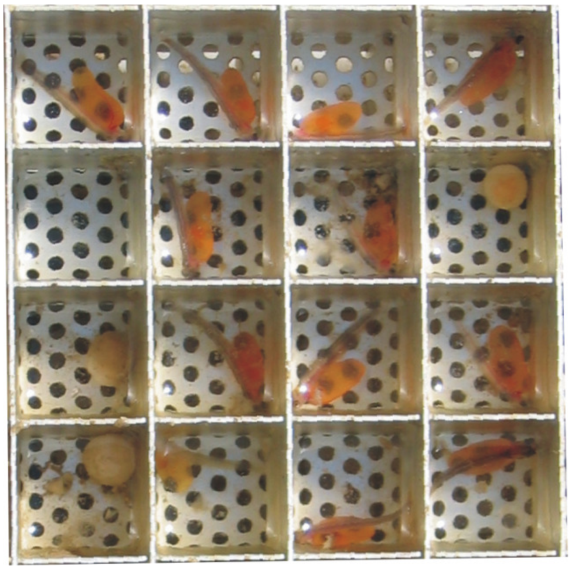


Fig. 4.2: Schematic side view of the exposed “egg sandwich” in the stream bed; note that the egg exposure unit is situated upstream of the water quality measurement unit.

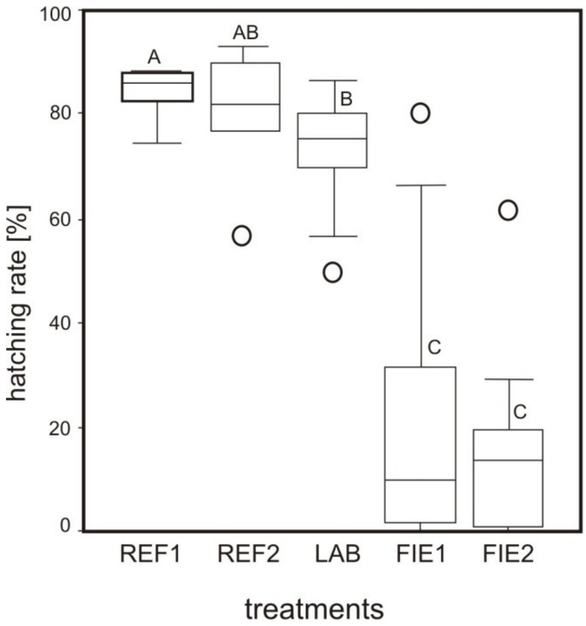
4.4 Results

The results of these comparisons (Fig. 4.4) revealed that HR of eggs exposed in the ES (mean ± s.d. = 80 ± 13%) did not differ (Tukey HSD, $p > 0.05$) from those in the upflow incubation trays (84 ± 5%), suggesting no systematic correction factor must be applied when using an ES exposure instead of direct egg exposure. Hatching periods in the ES closely (± 2 days) matched the hatching periods of the reference samples. Mean HR did not differ ($p > 0.05$) between ES and WV-box in the field exposure, indicating that an assessment of stream substratum conditions will deliver similar results in both cases. Variability in HR was higher in the ES compared to the WV-box, however, since HR and physicochemical water variables differed markedly in different depth horizons within ES boxes (Fig. 4.5). In the field test set, concentrations of dissolved oxygen, nitrite and nitrate, redox potential and pH value were the most determining factors for egg survival, whereas temperature, concentration of ammonium and electric conductivity explained little, or no variation in hatching success. Considering the significant sampling site effect ($p < 0.001$) on HR in ANOVA with interactions, the relation between exposure depth and hatching success became significant ($p < 0.001$). This indicates that an assessment of habitat quality at a high spatial resolution on a microhabitat scale is advantageous. Hatching rates in the laboratory flume resembled those of the reference exposures in the upflow incubation trays but were significantly lower in both field exposures. This result can most probably be explained by the adverse effects of high fine sediment loads and low oxygen values in the stream substratum of the River Moosach compared to the flume exposure in coarse substratum with high oxygen saturation. Thus, the salmonid egg development in the ES unit is not significantly different from that under natural conditions if water quality is sufficient, which proves the suitability of the ES for assessing stream substratum quality.



a	a	a	a
d	a	a	c
c	a	a	a
c	b	a	a

Fig. 4.3: Proposed evaluation key for the "egg sandwich"; assessment of salmonid egg development (a = living fry, b = dead fry, c = undeveloped egg, d = chamber empty).



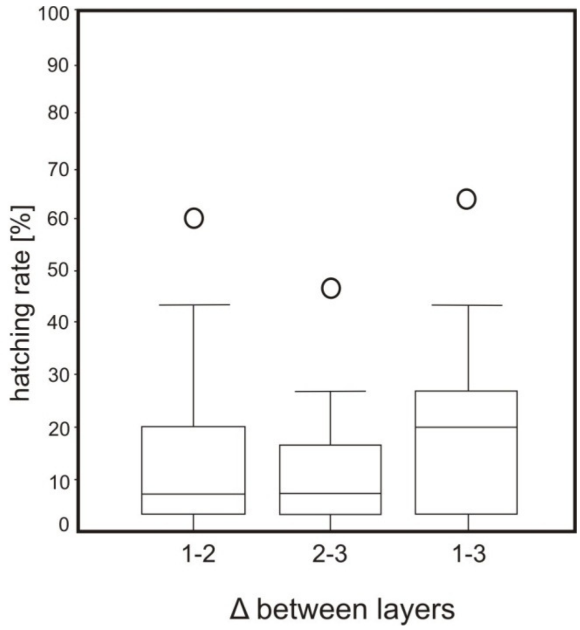


Fig. 4.4: Left. Comparison between different egg exposure treatments: REF1 = reference exposure in upflow incubation trays under regular hatchery conditions (n = 4 replicates, each with 1,000 eggs), REF2 = reference exposure in the “egg sandwich” under the same conditions as REF1 (n = 6 replicates, each with 30 eggs), LAB = “egg sandwich” exposure in substratum within a lab flume (n = 18 replicates, each with 3 x 30 eggs at depths 1,2,3), FIE1 = “egg sandwich” exposure to natural stream substrata in the river Moosach (n = 25 replicates, each with 3 x 30 eggs at depths 1,2,3), FIE2 = Whitlock-Vibert box exposure (encased with 1mm gauze to avoid the escape of fry) at the same sites like FIE1 (n = 25 replicates, each with 200 eggs); different letters indicate significant differences at $p < 0.05$; box plots show the median and the interquartile range, circles represent outliers.

Fig. 4.5: Right**.** Pairwise differences in hatching rates between three different depth layers of stream substratum exposed “egg sandwich” boxes in the river Moosach (n = 25 for each depth layer); note that Δ is greatest between the most distant depth layers 1 and 3, although overall differences are not significant (ANOVA, p = 0,066).

4.5 Discussion

Different alternative systems for hatching salmonid eggs in streams have been previously described (Vibert 1949, Whitlock 1979). Most of these systems, however, were primarily designed for salmonid propagation with the purpose of directly releasing hatched fishes into the stream. These systems are of limited use for assessing HR and for linking these with stream substratum quality variables, although some authors describe the use of modified Whitlock- Vibert boxes suitable for assessing hatching success (Mackenzie & Moring 1988). Systems specifically designed for the assessment of HR under natural conditions both during exposure to the free-flowing water (Rubin 1995, Donaghy & Verspoor 2000) and in the stream substratum (Harris 1973, MacCrimmon et al. 1989, Pauwels & Haines 1994, Rubin 1995, Donaghy & Verspoor 2000, Bernier-Bourgault et al. 2005, Dumas & Marty 2006) have been developed. Most of these methods, however, have not been designed to allow an assessment of spatial variation at different substratum depths (Harris 1973, Pauwels & Haines 1994, Rubin 1995, Bernier-Bourgault et al. 2005), which appears to be crucial at least in the stream investigated in this study. As far as is known, however, none of these systems is coupled with a measurement unit, which allows linking the biological effect of hatching success with adjacent water variables. Also, exposure of single eggs in separate chambers is more difficult with other systems compared to the ES, which can be a crucial factor if infection and transmission of fungi is a major problem. Due to the compact slight design of the ES and the planting technique of creating a small gap in the riverbed substratum, the disruption of the interstitial zone is marginal compared to the planting of other systems (Donaghy & Verspoor 2000). In conclusion, practical experience with the use of the ES suggests that this technique provides an easy tool with high operational reliability for assessing stream substratum quality by linking spatially resolved salmonid egg survival and physicochemical water variables from the same sites within the interstitial zone. This system may also be used for incubation of other species, such as juvenile freshwater bivalves, for which assessment of stream substratum quality is of great importance (Buddensiek et al. 1990, Geist & Auerswald 2007).

5 Seasonal and spatial bank habitat use by fish in highly altered rivers – a comparison of four different restoration measures

J. Pander & J. Geist

Ecology of Freshwater Fish 2010, **19**, 127-138

5.1 Abstract

River regulations have resulted in substantial modifications of the characteristics and the diversity of stream ecosystems. Fish habitat use in the context of species life histories and temporal habitat dynamics are crucial for the development of sustainable measures of habitat restoration in degraded rivers. The objective of this study was to compare the effects of introducing four different in-stream structures (bank ripprapp, benched bank ripp-rapp, successively grown riparian wood and artificial dead wood, nine replicates each) on the seasonal fish community distribution in a heavily modified stream ecosystem. Species richness and diversity, fish biomass and density showed strong variation (i) between habitat types, (ii) among replicates of the same habitat type, and (iii) in different seasons. The current low abundance of historical widespread rheophilic and migratory fish species in the study stream suggests that technical bank habitat restoration measures are only of limited use for the restoration of highly specialised target species in conservation such as *Barbus barbus* and *Chondrostoma nasus*. However, introduction of particular artificial stream structures (in particular of artificial dead-wood fascines) was found to concentrate biomass and density of none-specialised fish species like *Squalius cephalus*, *Alburnus alburnus*, *Gobio gobio* or *Rutilus rutilus*.

5.2 Introduction

Rivers and streams have been altered by mankind over centuries and are considered the most heavily modified ecosystems with an overproportional loss of biodiversity (Pimm et al. 2001, Gleick 2003). More than 70% of the large rivers of Europe, North America and the former Soviet Union are strongly regulated and there are more than 800,000 dams world wide (Dynesius & Nilsson 1994, Rosenberg 2000). The regulation of rivers caused by introduction of dams and weirs resulting in modification of natural flow regimes, as well as habitat fragmentation have all contributed to the decline of riverine fish species (Rosenberg et al. 1997, Aarts et al. 2003). Whilst restoration has attracted huge financial investment in recent times, there has been little or no consensus to date as to what constitutes successful ecological restoration (Giller 2005, Palmer et al. 2005). Knowledge of fish habitat use in the context of stream restoration and habitat rehabilitation measures is crucial for the improvement of their effectiveness. Scientific studies on the effects of restoration measures have mostly focused on large-scale restoration of complete streams or stream sections (e.g., Sear 1994, Jungwirth et al. 1995, Erskine et al. 1999, Kondolf et al. 2007, Hauer et al. 2008). However, results of these studies often cannot be applied to habitat rehabilitation within smaller sections of highly modified waterbodies (HMWB), where economical, technical and safety reasons (e.g., flood protection and hydropower use) rule out any large-scale restoration measures. In such cases, habitat restoration is usually limited to small-scale modification of bank habitats. The objective of this study was to investigate the seasonal fish habitat use in a highly modified model stream ecosystem, the river Günz (Germany) and to compare the effects of four different bank habitat modifications on seasonal distribution of the fish community. The Günz represents an ideal model system for this study as its size, geomorphology and flow regime are typical for many other rivers, which have become anthropogenically modified by their use for hydropower generation. In addition, the fish species composition in the Günz and the occurrence of target species for conservation such as *Chondrostoma nasus* and *Barbus barbus* were additional criteria for its selection as a study stream.

5.3 Material and Methods

Study area

The river Günz has a length of 55 km and a catchment area of 710 km^2. It arises out of the confluence of the western and eastern Günz and discharges into the Danube river. The downstream sections of the Günz are highly modified, mostly resulting from its use for hydropower generation. The study section resembles an artificial channel and is located between the two hydropower plants Ellzee (coordinates: 48°19´57´´N, 10°19´09´´E; construction year 1955) and Wattenweiler (coordinates: 48°18´49´´N, 10°19´51´´E; construction year 1945), about 16 km upstream the river mouth in Günzburg (Fig. 5.1). It has a total length of 2.45 km, and a hydraulic gradient of 0.0161%. The mean annual discharge measured at the nearest water gauge (Waldstetten, 48°21´09´´N, 10°18´08´´E) is about 8.35 $m^3 \cdot s^{-1}$ and ranges from 3 to 111 $m^3 \cdot s^{-1}$ (data available at http://www.hnd.bayern.de). Water temperatures vary between 0 °C in winter to 21 °C in summer (Wasserwirtschaftsamt Augsburg, unpublished data). Due to the lack of fish passes, upstream migration of fish is impossible and limited downstream migration is only possible during high flow conditions.

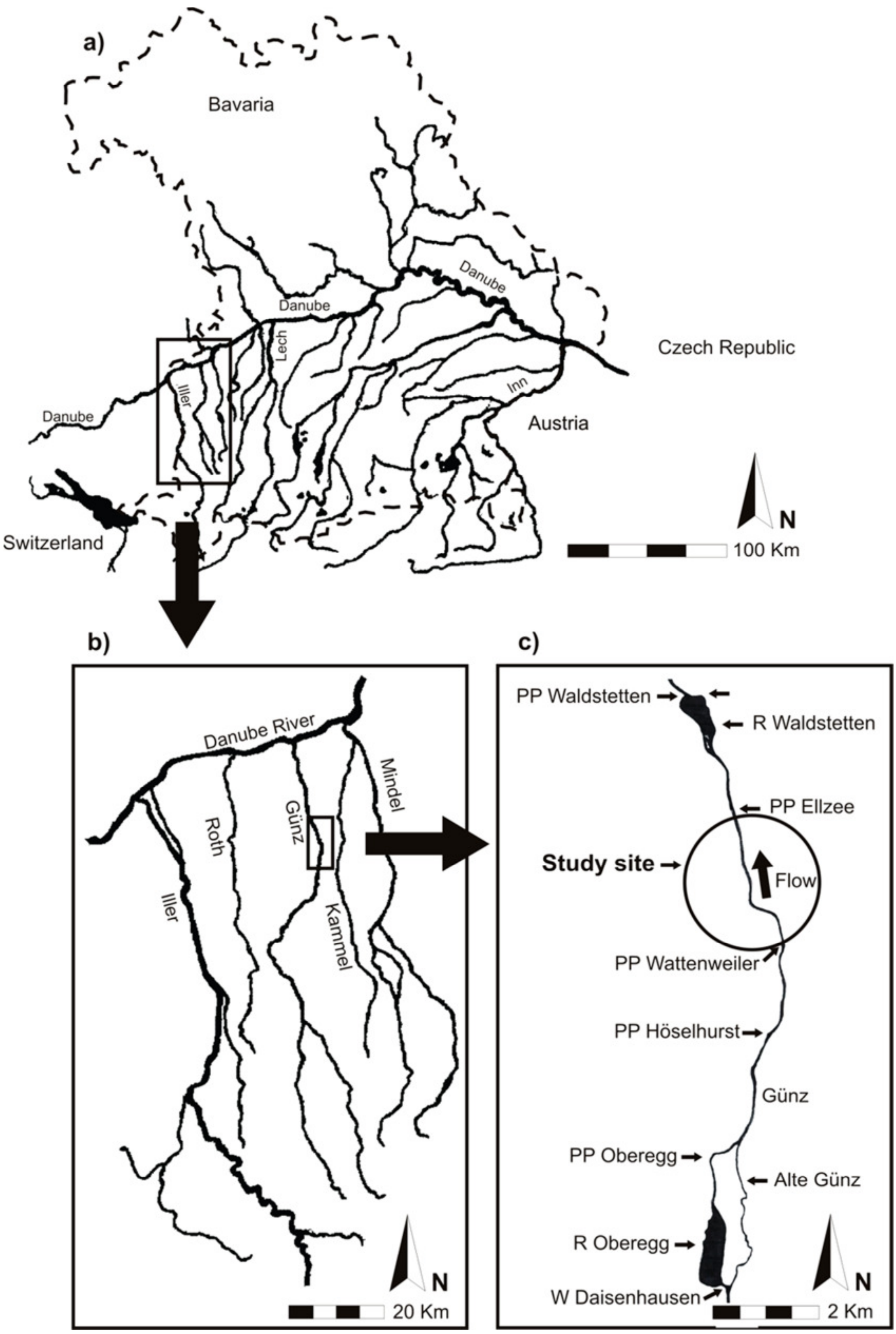


Fig. 5.1: Map and magnification of the study area with a) the upper Danube drainage system in southern Bavaria (Germany), b) the drainage area of the river Günz with its two tributaries and adjacent rivers, c) the flow channel of the study site with the lower and upper power plants (PP), reservoirs (R) and weirs (W).

Comparison of bank habitat types

Within the study section, the effects of four different bank habitat restoration measures on the fish community were compared in winter and summer season 2008 (2 years after they were built). The in-stream bank habitat rehabilitation included the introduction of dead wood, the introduction of shallow water zones, the introduction of boulders with different void sizes between them, as well as the maintenance of overhanging bank vegetation versus clearcutting (Fig. 5.2). The planning and construction of the habitat restoration measures were carried out by the hydroelectric power plant company in winter 2005 / 2006. Nine replicates of each of the four habitat types were introduced, resulting in a total of 36 study segments.

These 36 study segments represent all available bank habitat types in this section. To avoid the introduction of any systematic bias due to the sampling design, replicates were evenly distributed throughout the study section and had a mean distance of 100 m between them. The first 200 m downstream of the turbine run-off of the power plant was excluded to avoid the introduction of any bias into the dataset due to the lower water depth and higher flow velocity at the turbine outlet. The investigated bank length of the study segments was standardised to 30 m, as previously suggested by other authors (e.g., Grossman et al.1987, Grossman & De Sostoa 1994a). In total, 1080 m of bank habitat out of 4900 m total bank habitat length (22%) in the study section was assessed. Habitat HA comprises a combination of boulder bank enforcement with overhanging successively grown riparian wood and an understorey of shrub vegetation. Habitat HB consists of the same bank reinforcement as HA but has overhanging shrubs as bank vegetation. The characteristics of habitat type HB with frequent clear-cutting (spring and fall) of overhanging riparian wood vegetation, as well as the geomorphology and bolder size closely resemble the overall study section of the Günz before restoration and can thus be considered close to a reference control. Habitat HC consists of smaller boulders and is constructed with a 1.0- to 1.5-m wide berme with a shallow water area of 5–25 cm depth. The embankment is covered by grassland and overhanging shrub vegetation is absent. Habitat HD is structurally similar to HC but additionally comprises an artificial deadwood fascine (anchored by steel wires) with a diameter of 1.0-1.2 m and a length of 15.0 m. Spaces between the branches of the fascine are large enough to serve as refuge or habitat for smaller fish.

The discharge of the river Günz during the winter and summer sampling was 5.0 and 5.5 $m^3 \cdot s^{-1}$, respectively. Water depth, flow velocities at the surface and above the stream bed, water temperature, oxygen content, pH-values and electrical conductivities were measured at each sampling date at each of the 36 sampling sites. Generally, a great similarity of these variables among the different habitat types was observed.

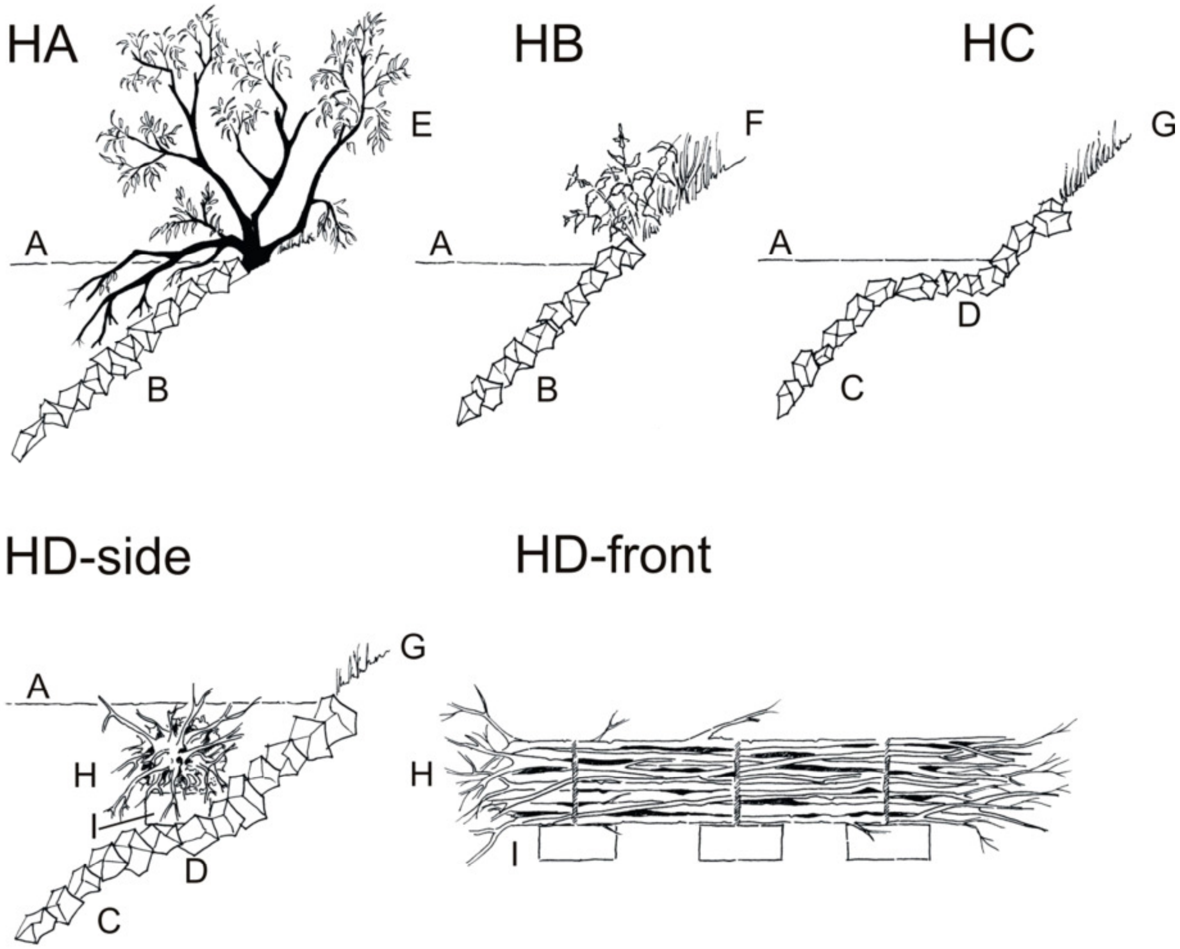


Fig. 5.2: Schematic of the four assessed habitat types with: HA=bank reinforcement with overhanging riparian wood; HB=bank reinforcement with overhanging shrubs; HC=benched bank reinforcement with graminaceous vegetation; HD-side=side-view of benched bank reinforcement with dead wood; HD-front=front-view of the dead wood; A=water surface level, B=boulder ripp-rapp with limestone 40 cm to 80 cm size, C=boulder ripp-rapp with limestone 20 cm to 40 cm size, D=benched bank reinforcement, E=natural grown riparian wood, F=natural grown shrubs, G= sowed grassmixture, H=artificial made dead wood fascine, I=concrete block to anchor the fascine.

Fish sampling

The fish community was assessed on 14 February 2008 and on 19 June 2008 using a boat based electrofishing generator (EL 65 II; Grassl, Schoenau, Germany). The study segments were consecutively sampled with the same electrofishing crew within a 5-h period (10 a.m.–3 p.m.) working from downstream to upstream direction. A single anode was used and stunned fish were collected with a dipnet while the boat was driving upstream at a constant distance of 3 m to the bank. All samples were taken along the bankside of the boat. The electrofishing time per study segment was 5–8 min, resulting in an average sampling speed of 0.06–0.10 m s^{-1}. Fish from each replicate were held in separate plastic tanks with oxygen supply. The total length of all specimens was measured to the nearest cm. Fish of 10 cm or more were individually weighed to the nearest gram. For smaller specimen, a representative number of at least 15 fish was weighed to determine the body mass index, BMI = [weight (g) / total length^3 (cm)] · 100, and to determine the total biomass. The same methodology was used at both sampling dates. All fish were directly released at the sampling sections from which they had been collected. Mortality before release of the fish was only detected at the first sampling date at one single sampling site (HD) in one species, bleak (*Alburnus alburnus*).

Statistical analysis

A comparison of seasonal fish habitat use in the four different habitat types was based on direct comparisons of fish species richness and fish biomass. In addition, fish diversity distribution measured by maximum diversity index per habitat type (HMAX), α-diversity (Shannon & Weaver 1949) and Evennes (Pielou 1966) were computed. β-diversity was calculated as species turn-over (βt) as originally suggested in Wilson & Shmida (1984) with minor modifications (Beierkuhnlein 2003, Koleff et al. 2003). Normality of data was tested with the Kolmogorov–Smirnov test and the homogeneity of variances was tested with the Levene test. As most data / residuals were normally distributed, one-way analysis of variance (ANOVA) and – in case of significance – Tukey post hoc tests were carried out to test for differences in mean values between the four habitat types. If data or residuals were not normally distributed Kruskal–Wallis ANOVA and in case of significance Wilcoxon signed-ranks post hoc tests were performed. Pairwise seasonal comparisons of habitat types were carried out using a paired samples t-test [testing variables were species richness, numbers of specimen, biomass and mean fish weight (MFW)]. All statistical analyses were carried out using SPSS (SPSS version 11.0, SPSS inc., Chicago, Illinois, USA). Significance was accepted at $p \leq 0.05$. Bonferroni corrections were applied for multiple tests. Additionally, multi-dimensional scaling (MDS) analysis using the species richness as input variable was used for a comparison of habitat types. The summation of caught species of each habitat replication was transformed into similarity values by calculating the differences between the number of species of all habitats assessed (De Leeuw & Mair 2009). The MDS was carried out using the SMACOF package of the open-source program R (http://www.r-projekt.org).

5.4 Results

Fish habitat use

Over all habitat types and sampling sites in the river Günz, 20 fish species from seven families were found. The fish community was highly dominated by cyprinids, comprising 14 species. Only five species were typical rheopar or reophil (*Cottus gobio*, *Barbatula barbatula*, *Barbus barbus*, *Chondrostoma nasus* and *Gobio gobio*) whereas the other species can be described as indifferent, euryopar or limnophil (following the habitat classification by Zauner & Eberstaller 1999). The reophilic target species *Barbus barbus* and *Chondrostoma nasus* were only rudimentary present in terms of number of individuals and biomass (Fig. 5.3). Strong differences in fish biomass and in the number of individuals were detected between the four different habitat types, whereas these differences were less pronounced for the number of species (Table 5.1). The highest species richness was found in HD with 17 species (ranging from 8 to 12 within individual replicates; SD = 1.6), followed by HA (16), HB (15) and HC (14). Mean species richness was significantly different for the pairwise comparisons of HD to HA (Tukey HSD, $P < 0.001$), HD to HB (Tukey HSD, $P < 0.001$) and HD to HC (Tukey HSD, $P = 0.011$). Species richness among all replicates ranged from 2 (one replicate of HA, HB and HC) to 12 species (three replicates of HD). In the dead-wood fascine habitat HD, 94.8% of bleak (*Alburnus alburnus*), 82.4% of roach (*Rutilus rutilus*), 80.8% of gudgeon (*Gobio gobio*) and 64.0% of minnow (*Phoxinus phoxinus*) were caught.

		a	b			c	d	e	f	g	h	i
Habitat type		Species	M	±SD	Mdn	Total [g] [min-max]	Total [min-max]	M[g]	HS	HMAX	HS/HMAX	β_t
HA	Summer	15	3.9	1.7	3	37,853 [170 - 16,364]	163 [3 - 42]	232 [0.7 - 6,770]	1.91	2.71	0.70	0.94
	Winter	7	1.8	1.8	1	43,501 [0 - 31,605]	184 [0 - 152]	236 [0.3 - 1,605]	1.00	1.95	0.51	
HB	Summer	13	3.4	2.3	2	10,880 [553 - 2,115]	121 [8 - 21]	90 [1.2 - 1,530]	1.19	2.56	0.46	0.85
	Winter	10	2.9	2.0	3	4,482 [0 - 3,619]	85 [0 - 24]	53 [1 - 631]	1.72	2.30	0.75	
HC	Summer	13	4.3	2.0	4	2,866 [93 - 704]	197 [7 - 47]	15 [0.7 - 147]	1.59	2.56	0.62	0.82
	Winter	11	4.2	2.9	5	1,593 [1 - 1,128]	158 [1 - 68]	10 [0.3 - 980]	1.83	2.40	0.76	
HD	Summer	16	7.7	2.2	8	15,904 [210 - 4,260]	671 [5 - 147]	24 [0.3 - 440]	1.84	2.77	0.66	0.80
	Winter	13	6.1	1.5	6	15,148 [163 - 8,611]	2,268 [10 - 1,587]	7 [0.8 - 1,050]	0.53	2.56	0.21	
Overall	Summer	19	4.8	2.6	4	67,503	1,152	59	2.03	2.94	0.69	0.72
	Winter	15	3.8	2.6	4	64,725	2,695	24	0.98	2.71	0.36	
	All	20	4.3	2.6	4	132,228	3,847	34	1.64	3.00	0.55	

Table 5.1: Summary of seasonal fish species composition, biomass and diversity at the four different habitat types of the river Günz: a) total number of species; b) mean number of species (M), median of the species number (Mdn), standard deviation of species number (±SD), N= 9 replicates; c) upper lane: fish biomass, lower lane: range with minimum and maximum; d) upper lane: number of fish, lower lane: minimum and maximum number of fish; e) mean fish weight; f) Shannon-Index for diversity (HS); g) maximum diversity index per habitat type (HMAX); h) Evennes (HS/MAX) and i) Wilson & Shmida species turn-over index (βt). HA, HB, HC, HD refer to the four habitat types as described in Figure 5.2.

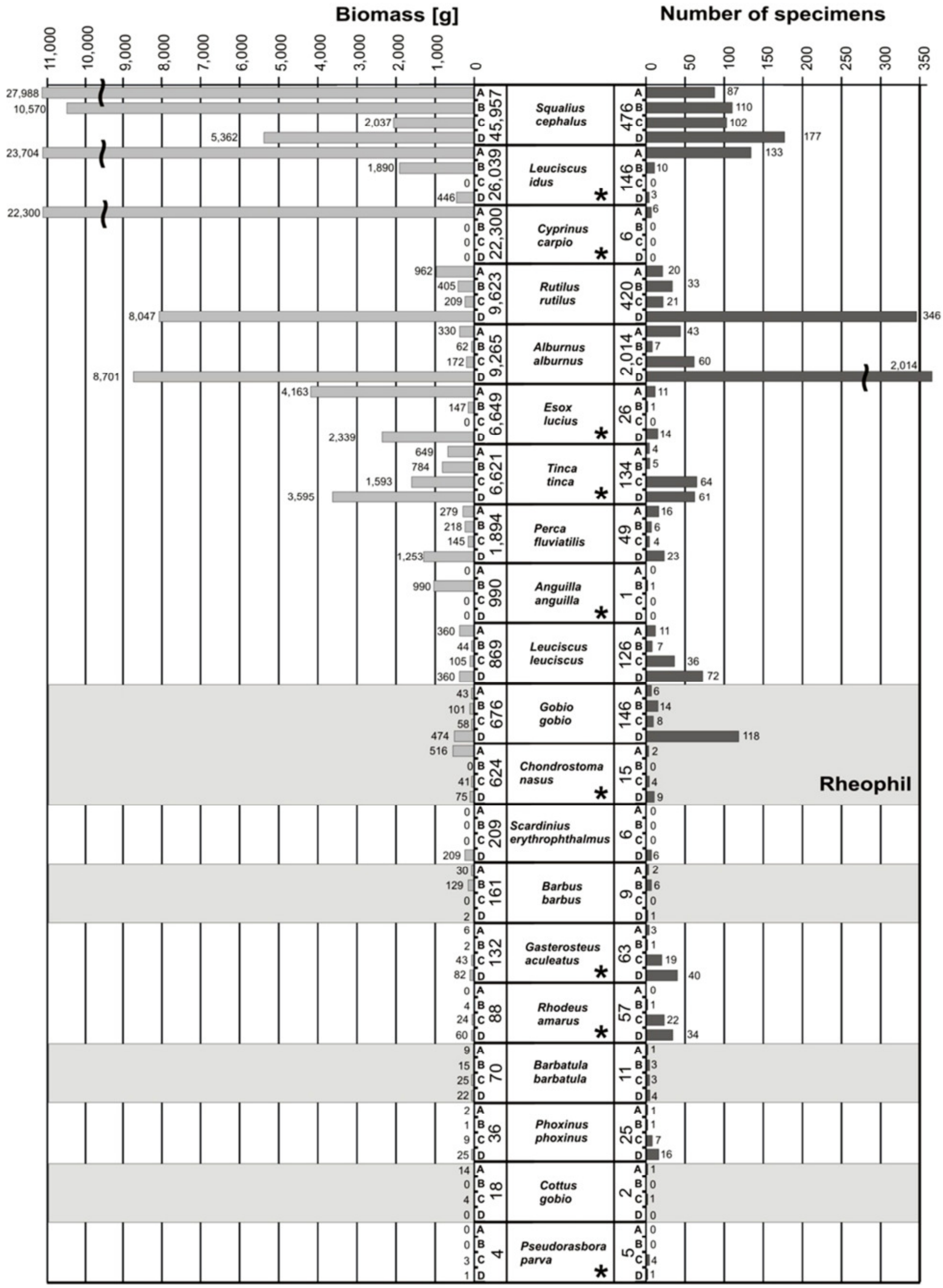


Fig. 5.3: Cumulative fish biomass and number of specimens in the four different habitat types HA (A), HB (B), HC (C) and HD (D; n=18 replicates, 2 seasons x 9 habitats for each habitat type). Species are arranged according to their total biomass contribution; * indicates alien or stocked species; rheophilic species are highlighted in grey.

Species diversity measured by the maximum diversity index per habitat type (HMAX) was highest in HD (Fig. 5.4). This ranking prevailed during summer and winter sampling whilst highest values for HMAX were consistently detected during the summer sampling (Table 5.1). Considering the abundance of species in the Shannon index, a similar pattern was observed.Seasonal fluctuations of diversity calculated by the Wilson & Shmida species turnover index was significantly higher in HA compared to all other habitat types.

A total number of 3847 fish were caught during the study, with 2939 fish in HD, 355 in HC, 347 in HA and 206 in HB. Despite the strong total difference, mean fish counts did not differ significantly among habitat types due to the strong variation introduced by highly differing numbers of *Alburnus alburnus* among replicates. Excluding *A. alburnus* from the dataset, the differences in mean numbers of fish specimen became significant for the pairs HA–HD (Tukey HSD, P = 0.004), HB–HD (Tukey HSD, P = 0.003) and HC–HD (Tukey HSD, P = 0.012). In all habitat types, indifferent species contributed significantly to total fish abundance (Fig. 5.3). Main species in HD were *Alburnus alburnus* (69% of all individuals within this habitat type), *Rutilus rutilus* (12%) and *Squalius cephalus* (6%), in HC *Squalius cephalus* (29%), *Tinca tinca* (18%), *Alburnus alburnus* (17%), in HA *Leuciscus idus* (38%), *Squalius cephalus* (25%) and *Alburnus alburnus* (12%), and in HB *Squalius cephalus* (53%), *Rutilus rutilus* (16%) and *Gobio gobio* (4%).

The total fish biomass caught in the four habitat types was 132.2 kg. Among habitats, the total fish biomass varied by a factor of 20 (Table 5.1) with the highest value in HA (81.4 kg), followed by HD (31.1 kg), HB (15.4 kg) and HC (4.5 kg). The same indifferent species which dominated fish abundance were also most important in terms of their biomass contribution (Fig. 5.3). The mean individual fish weight varied by a factor of 20 among habitat types with the biggest fish found in HA (234 g), followed by HB (75 g), HC (13 g) and HD (11 g). These differences were significant for the pairs HA–HB (Wilcoxon signed ranks test, P = 0.023), HA–HC (Wilcoxon signed-ranks test, P = 0.002) and HA–HD (Wilcoxon signed ranks test, P = 0.002).

The similarity of the four bank restoration measures is visualised in the MDS analysis (Fig. 5.5). The strongest differentiation of the two habitat types HD and HA is in concert with the strong difference between HD and the other habitat types in terms of fish density, fish biomass, MFW and seasonal abundance of species. This finding is remarkable, as HD was introduced as a surrogate restoration measure with the original intention of replacing the functionality of HA.

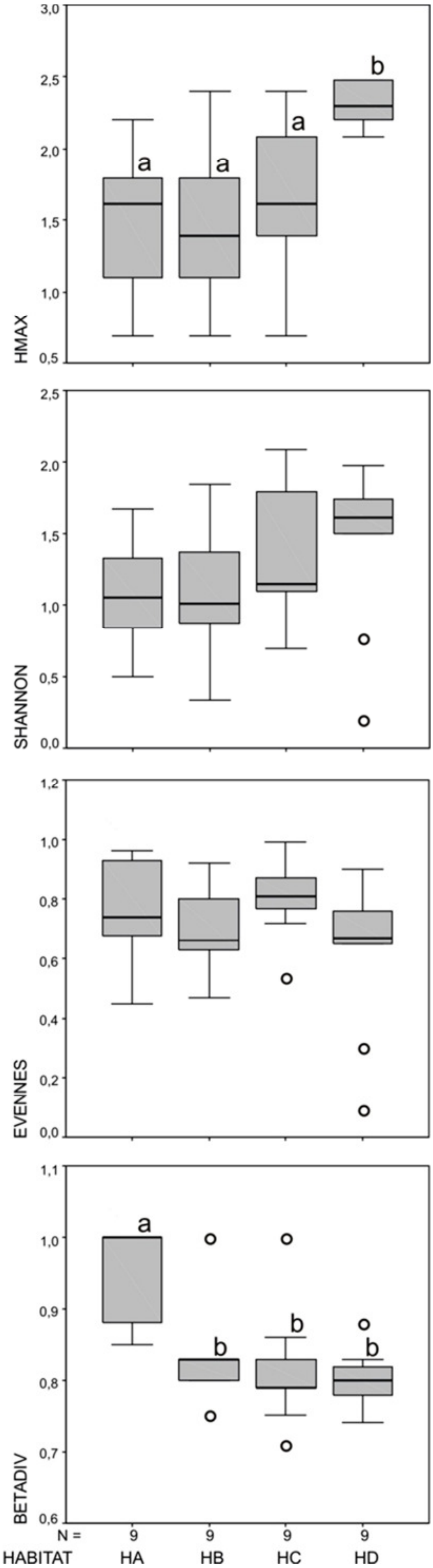


Fig. 5.4: Characterization of biodiversity in the four habitat types (combining summer and winter sampling): maximum diversity index per habitat type (HMAX), Shannon-Index for diversity (SHANNON), Evennes, and Wilson & Shmida species turn-over index (BETADIV). HA, HB, HC, HD refer to the four habitat types as described in Figure 5.2; box: 25% quantile, median, 75% quantile; whisker: minimum, maximum values; circles: outliers (3.5 x SD); different letters indicate significant differences at $p < 0.05$.

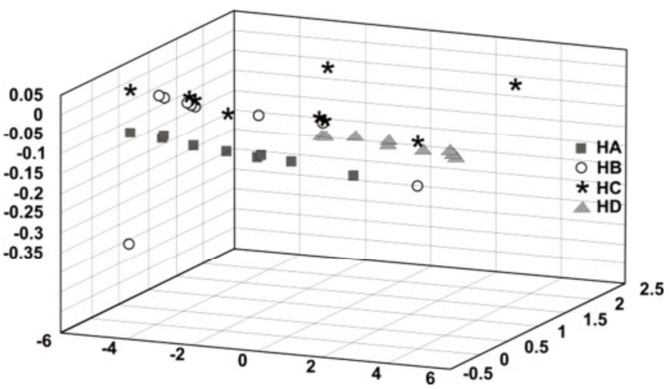


Fig. 5.5: Multi dimensional scaling analysis (MDS) of the four habitat restoration measures considering species richness as input variable. All axes showing similarity values between habitats three- dimensional. The MDS calculation results in a stress of 0.00032 which indicates a high reliability of the clustering and distances between habitats. HA, HB, HC, HD refer to the four habitat types as described in Figure 5.2.

Seasonal effects

Strong seasonal effects on species richness were detectable (paired samples test, $P < 0.001$). However, due to the strong variability among replicates, no significant differences in mean values between the sampling dates were found for the number of specimens, biomass (summer 51% and winter 49%) and MFW.

A smaller number of species was found during winter (15) compared to the sampling at summer time (19). Five species (*Barbus barbus*, *Cyprinus* *carpio*, *Pseudorasbora parva*, *Cottus gobio* and *Anguilla anguilla*) were only caught during summer sampling. The rheophilic target species *Chondrostoma nasus* (15 specimens) was only detected during the winter sampling. The species turnover rate varied from 0.798 to 0.940 and was highest in HA (Fig. 5.4, Table 5.1). Each habitat type showed a higher number of species in summer compared to the winter sampling with the greatest differences in HA (eight species), followed by HB and HD (three species) and HC (two species). The differences due to the seasonal sampling were highly significant in HA (paired samples test, $P < 0.001$) and HD (paired samples test, $P = 0.003$). Habitats HB and HC showed no significant difference between summer and winter sampling.

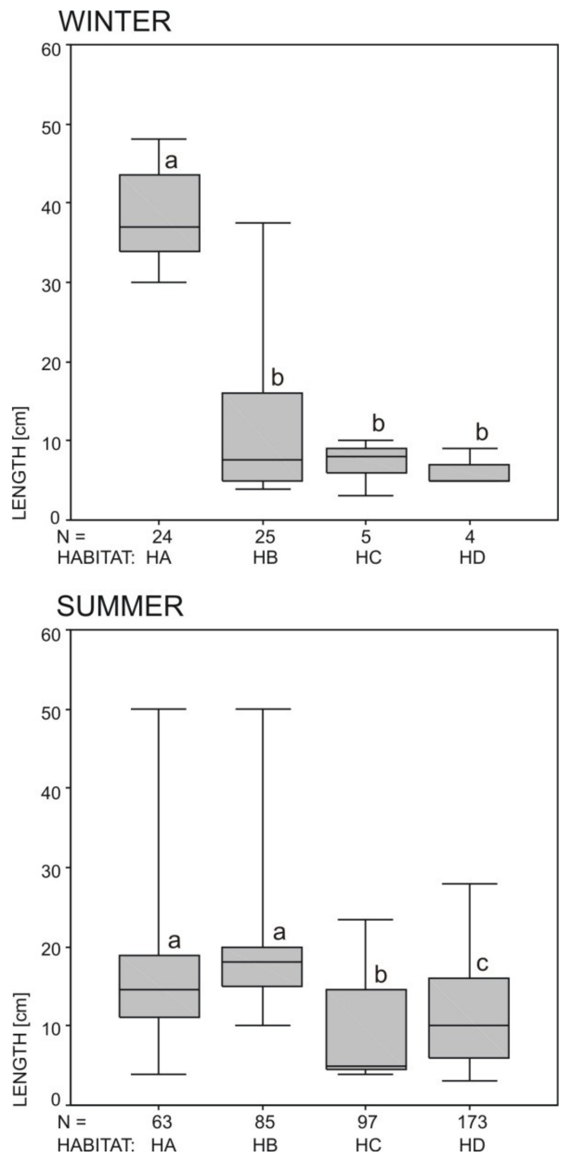


Fig. 5.6: Size-dependent seasonal habitat use of chub (*Squalius cephalus*), the dominant fish species in the study section. HA, HB, HC, HD refer to the four habitat types as described in Figure 5.2. N refers to the number of chub detected per sampling; box: 25% quantile, median, 75% quantile; whisker: minimum, maximum values; different letters indicate significant differences at $p < 0.05$.

Seasonal fluctuation in species diversity measured by the Shannon index varied by a factor of 3.5 between the two sampling dates in HD. These differences were less pronounced for the other habitat types (HA = 1.9, HC = 0.87, HB = 0.69). The Evennes index indicates that HD is the most uneven habitat type in winter (Table 5.1).

Different numbers of specimens were caught at the two sampling dates, with 30% of individuals caught during summer and 70% during the winter sampling, despite the fact that the number of fish in HA, HB and HC only showed marginal differences between 2% and 5%. Major seasonal differences were found in HD, where more than 76% of all sampled individuals were caught (17% in summer and 59% in winter).

Remarkably constant catches of total fish biomass were recorded at the two sampling dates (68 and 65 kg at the summer and winter sampling, respectively). Seasonal variability of fish biomass in the four habitat types varied by factors of 2.43, 1.80, 1.05 and 0.87 for HB, HC, HD and HA, respectively, indicating seasonal movements of fish between them and / or other habitats of the main channel. Also, a shift in the contribution of single species to biomass in the habitat types was observed. *Squalius cephalus* and *Leuciscus idus* were the two major species in terms of biomass in HA during winter but were replaced by *Cyprinus carpio* during summer. The high biomass contribution of *Alburnus alburnus* in HD in winter was replaced by a multiple increase of nearly all other species caught in this habitat type in summer, mainly *Squalius cephalus* and *Rutilus rutilus*. The MFW differed between seasons in HD (factor 3.4), HB (factor 1.7), HC (factor 1.5) but showed no difference in HA (factor 1.0). Fish appeared to be more aggregated during winter sampling compared to summer sampling, as evident from the absence of fish in 4 out of 36 replicates (3 in HA, 1 in HB; 11%) during winter sampling and no single replicate without fish during the summer sampling.

The size-dependent seasonal habitat use of the dominant fish species, *Squalius cephalus*, indicates that larger (adult) fish mostly occurred in habitat types HA and HB (Fig. 5.6). During summer sampling, HA and HB held chub in a size range of up to 50 cm, whereas the maximum sizes did not exceed 23 and 28 cm in HC and HD, respectively. The relative differences in fish size and density were even more pronounced during winter sampling with a complete lack of chub smaller than 30 cm in HA but no fish larger than 10 and 9 cm in HC and HD, respectively. Overall, the number of chub in all four bank habitat types was sevenfold higher in summer (n = 418) compared to winter (n = 58).

Roach (*Rutilus rutilus*) had a clear preference of habitat HD with 65% (winter) and 96% (summer) of the individuals being caught near the dead-wood fascine. As previously described for chub, the largest specimens of roach were also found in HA during winter time.

5.5 Discussion

Fish habitat use

Knowledge of fish habitat use in the context of stream restoration and habitat rehabilitation measures is crucial for the improvement of their effectiveness.

However, only few studies have assessed the effects of bank habitat restoration in highly modified water bodies. Studies about the effects of large-scale restoration measures have improved the understanding of their effectiveness (Wesche 1985, Jungwirth et al. 1995, Stanford et al. 1996, Ward et al. 2001, Hauer et al. 2008), but they often cannot be applied to habitat rehabilitation within smaller sections of HMWB, where economical, technical and safety reasons rule out any large-scale restoration measures. The novel topic in this study is the assessment of small-scale bank habitat restoration measures in a HMWB to compare their effects on the fish community, using nine replicates of four different restoration measures.

The catch of 20 fish species found in this study shows that HMWB can serve as habitats for a remarkable number of fish species and that HMWB should thus be included in concepts of aquatic biodiversity conservation. Overall, the species caught in this comparative study of bank habitats seem to be representative of the overall species composition in the highly altered sections of the Günz in this area. Both catch data from local fishermen (F.J. Schick, personal communication) and data from electrofishing surveys (Müller, personal communication) in our study segment and the upstream and downstream sections revealed smaller or consistent numbers of fish species and indicate similar community structures.

The species composition found in the Günz cannot be assigned to the fish regions classification system proposed for natural streams (Sparks et al. 1990, Zauner & Eberstaller 1999). Instead, the fish community structure of the Günz strongly resembles a non-natural mix of species from different fish regions with a dominance of nonspecialist species from the potamal. Lateral and longitudinal dams result in a uniform and fragmented system with anthropogenically shortened bank habitat length. In this case an agglomeration of different fish regions between the power plants (1.8–2.4 km at the Günz) replaces a natural fish region sequence and can explain the enhanced species richness.

In fact, a high fish species richness attributable to nonspecialised species has been shown to be negatively linked with stream ecosystem health in headwater regions (Marchetti et al. 2004, Geist et al. 2006, Erös 2007) and is thus not always desirable in conservation management (Marchetti & Moyle 2001).

The low number of historically widespread reophilic specialists and the high density and biomass of ubiquitous fish species found in this study indicate that none of the four habitat restoration measures can compensate for the deficient habitat quality for rheophilic fish in this stream. In the Günz, originally abundant stenoecious fish like *Barbus barbus* and

Chondrostoma nasus only occur in relict populations within the HMWB. A successful reproduction of *Barbus barbus* and *Chondrostoma nasus* inside the study segment seems currently impossible due to the deficits of habitats for critical life stages of these species, which require particular spawning and juvenile habitats. Especially, juvenile habitats which can normally be found along river banks in areas with low current speed, fine coarse substratum and water depths to a maximum of 40 cm (Jurajda 1999, Schiemer et al. 2003, Hauer et al. 2008) are almost entirely absent in HMWB. The small numbers of *Barbus barbus* and *Chondrostoma nasus* caught suggest that self-sustaining minimum viable populations cannot be achieved for rheophilic specialists as long their main habitat requirements during critical phases of their life cycle are not met qualitatively and quantitatively.

The temporal distribution and aggregation of fish in different habitats are influenced by the dynamic and complex interactions between habitats and species in the context of their ecological niches. Main factors for the seasonal and spatial distribution of fish include food availability, structural aspects (hiding places, etc.), as well as population densities and competition and predation (Grossman et al. 1987, Grossman & De Sostoa 1994a und1994b, Prenda et al. 1997, Erös et al. 2003). All four of the restoration measures held high fish species richness but the aggregation of fish biomass in HA and of fish density in HD indicate that riparian or dead-wood textured habitats seem to be most attractive for fish. These structures were also most distinctive from all other habitats in the MDS analysis. Dead-wood accumulations provide a variety of habitats for many fish species simultaneously comprising hiding spots, feeding grounds and spawning habitats. Their importance in stream habitat restoration has been emphasised by other authors as well (e.g., Angermeier & Karr 1984, Piegy & Gurnell 1997, Grossman & Ratajczak 1998, Quist et al. 2005).

Seasonal effects

The strong seasonal effects on fish species richness and density indicate pronounced fish migrations between bank habitats and the main channel, despite the fact that the total biomass caught and its distribution among habitats remained constant over time. Variability in species richness between summer and winter sampling was most pronounced in HA where 15 species were detected during the summer sampling but only seven during the winter sampling. In accordance with recommendations by other authors (e.g., Nykänen et al. 2001, Yu & Peters 2003, Daugherty & Sutton 2005, Vlach et al. 2005, Heermann & Borcherding 2006), the highly variable species distribution and the fluctuating fish density suggest that an assessment of habitat rehabilitation measures should generally include these seasonal effects.

Especially at the winter sampling, the increase of fish density and the decrease of MFW in HD show that artificial dead-wood fascines are excellent winter habitats for smaller species like *Alburnus alburnus*, *Rutilus rutilus*, as well as for juvenile *Squalius cephalus* and *Leuciscus leuciscus.* Larger fish like adult *Cyprinus carpio*, *Esox lucius* and *Squalius cephalus* generally seem to prefer the structures of HA where the MFW was almost constant at the two sampling dates. An explanation for this effect may be found in the obvious structural differences between HA and HD. The artificial dead-wood fascines create many small spaces and cavities between the branches of HD, which are particularly well accepted by small schooling fish.

Restoration recommendations

The four introduced restoration measures differ in terms of their cost and feasibility. The installation of benched ripp-rapp or common ripp-rapps as bank stabilisation is common in most flow channels for hydroelectric power generation and is typically costly. Successively, grown riparian wood keeps restoration costs low but it takes several years until the restoration measure can work efficiently and at this time stage it often has to be clear cut in European HMWB for flood safety reasons. In line with HA, the benched ripp-rapp HC needs several years to reach its optimum functionality stage as the successive development of makrophytic vegetation in these shallow water zones is a slow process. The dead-wood fascine HD is effective immediately after its introduction into the channel and holds high species richness and fish density. The material of the dead-wood fascines is cheap and can even be a by-product of dam management measures. Steel cables and concrete attachments for its anchoring in the stream can be standardised and keep the costs low. In combination with other structures, it can enhance the patchiness and interlocking of different habitats wherever dead-wood management is essential due to safety reasons. Due to differences in the ecological niches of specialised species, a diversity and combination of in-stream structures is likely to be most beneficial for the sustainable management of fish biodiversity.

6 Ecological functions of fish bypass channels in streams: migration corridor and habitat for rheophilic species

J. Pander, M. Mueller & J. Geist

River Research and Applications 2011, online early, DOI: 10.1002/rra.1612

6.1 Abstract

The introduction of weirs into stream ecosystems resulted in modifications of serial continuity and in the decline of riverine fish species. Successful river restoration requires information on the ecological functionality of fish bypass channels which are considered an ecological improvement according to the EU Water Framework Directive. In this study, we compared the functionality of three nature-orientated fish passes as compensatory habitats and migration corridors for fishes. Fish passes differed significantly from upstream and downstream sites of the weirs, revealing higher current speed, lower water depth, smaller channel width and greater habitat variability. Following these structural differences, they provided key habitats for juvenile, small and rheophilic fishes which are typically underrepresented in highly modified water bodies. All fish passes were used as migration corridors, with increased fish movements during high discharge and at spawning periods. Since river stretches with high variability of current speed and water depth are scarce in highly modified water bodies, fish passes can play an important role as compensatory habitats and should thus be considered more intensively in habitat assessments and river restoration. Ideally, fish bypasses should mirror the natural discharge dynamics and consider all occurring fish species and sizes.

6.2 Introduction

River regulations have resulted in substantial modifications of the characteristics and the diversity of stream ecosystems (Pimm et al. 2001, Gleick 2003). Especially the introduction of dams and weirs for milling and hydroelectric power generation has resulted in modifications of natural flow regimes, habitat fragmentation, and ultimately in the decline of riverine fish species (Rosenberg et al. 1997, Aarts et al. 2003). In particular, the quality of habitats for juvenile fishes and for reproduction are considered the major problems for rheophilic freshwater fishes (Pander & Geist 2010). In central Europe, dams and weirs have mostly been constructed for hydropower generation, which is the most important renewable source of energy worldwide (Bratrich et al. 2004, Demirbas 2007). Hydropower is regarded as a clean energy source since its production does not emit greenhouse gases or air pollution (Demirbas 2007), although the flow modifications caused by hydropower may have major ecological impacts on aquatic as well as terrestrial systems (Bratrich et al. 2004, Renöfält et al. 2010).

Alterations of the fish community of many small rivers date back to the 17th century when many dams and weirs were built, but they also continue today as a means of CO2-neutral production of energy. At present, European rivers comprise a mosaic of up- and downstream sites of weirs that succeed each other in short geographical distances.

The serial continuity of river systems and a natural fish community are hydromorphological and biological quality elements in the European Water Framework Directive (WFD, European Parliament 2000). In the course of the implementation of the WFD, the German parliament proclaimed a legal basis which enforces the free passability of all migration barriers in rivers. At the same time, there are many attempts to restore important key habitats for desired species, or even for complete river systems. Consequently, information on the factors that determine the functionality of semi-natural fish passes, as migration corridors and compensation habitats, is crucial for the assessment of the ecological status and for the monitoring of the success of restoration.

The objective of this study was to assess the conservation value of stream habitat restoration by nature-oriented fish passes for the fish community. Specifically, the role of the fish passes as compensation habitats for rheophilic fish species and sensitive live stages was assessed. In addition, the role of the fish passes as migration corridors at different flow conditions was evaluated. All sites were located in spatial proximity to each other and were built with a nature-orientated construction scheme.

6.3 Material and Methods

Study area

The study area is located in the northeast of Bavaria in one of the major drainage systems of southern Germany, the River Main. All three investigated fish passes are in spatial proximity to each other and were located at the rivers Nassach (length about 29 km, catchment area 158 km^2) and Riedbach (length about 15 km, catchment area about 45 km^2), a tributary of the Nassach (Fig. 6.1). The mean annual discharge measured at the nearest water gauge for the Nassach (Römershofen, 50°04'34"N, 10°31'07"E) is about 0.884 m^3•s^{-1} and ranges between 0.094 m^3•s^{-1} and 30 m^3•s^{-1} (data available at www.hnd.bayern.de). Water temperatures vary between 2 °C in winter to 23 °C in summer (Wasserwirtschaftsamt Bad Kissingen, unpublished data).

The fish passes were built in the years 2003 (Nassach Rain-Mill, site 1), 2004 (Nassach Zinken-Mill, site 2) and 2008 (Riedbach Ried-Mill, site 3). They were built to compensate deficits caused by the historical channel alterations due to the settlements of three overdrift mills in the 17th century. All three mills were abandoned and there has been no reallocation to hydropower generation. However, the historic justified water rights, weirs and dams still exist. The design of all fish passes was a nature-orientated construction scheme with a high variability of water depth, current speed and natural structures like dead wood accumulations and stones. The channel slope of all three fish passes was adapted to the natural conditions of the surrounding terrain and the sinouosity (SI) of the water courses (site 1: SI = 1.97; site 2: SI = 1.85, site 3: SI = 1.87) was constructed accordingly and is typical for small sized rivers in this landscape. Species-specific habitat and migration requirements were not specifically considered during the planning of the fish passes.

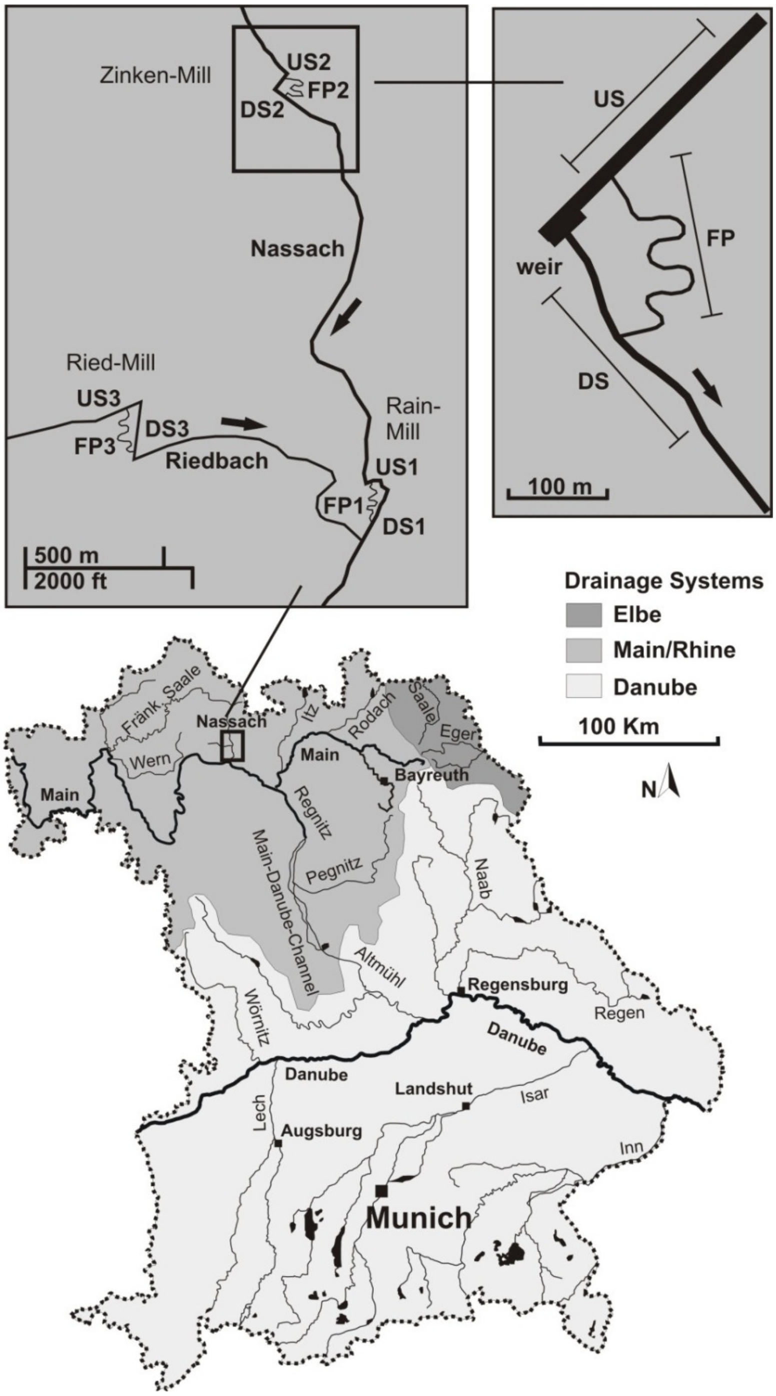


Fig. 6.1: The three major drainage systems of Bavaria (Germany) are printed in three different shades of grey on the base map. Detail views of the rivers Nassach and Riedbach and an exemplary detail view of a study site with fish pass (FP), upstream (US) and downstream (DS) sampling section. Black arrows indicate flow direction.

Fish community assessment

For the assessment of the functionality of the three fish passes as compensation habitats for rheophilic fish species, all fish passes as well as their adjacent up- and downstream sites were investigated. The length of the sampling area upstream and downstream was matched to equal the length of the associated fish passes, respectively (Fig. 6.1). Distances of 20 m directly upstream and downstream of the weirs were excluded from the sampling for safety reasons. For an assessment of the ecological integrity, biotic and abiotic habitat components were considered. Fishes are known as sensitive indicators for alterations of habitat conditions with short response times and are therefore a powerful tool for the assessment of the ecological integrity of the newly built semi-natural fish passes. For the assessment of free fish passability and migration directions, fish-traps were installed in all fish passes. To link the biotic fish data with abiotic habitat characteristics, additionally important physicochemical parameters were assessed at all sampling sites.

Fish sampling

The fish community was assessed in April 2009 using a electrofishing generator (EL 65 II, Grassl, Schoenau/ Germany). The discharge of the river Nassach and its tributary Riedbach during the electrofishing was 0.75 $m^3 \cdot s^{-1}$ and 0.35 $m^3 \cdot s^{-1}$, respectively. The study segments were consecutively sampled with the same electrofishing crew within one day per site working from downstream to upstream direction. All sites were blocked with two fishnet barriers (mesh size 6 mm) during the sampling to avoid the escape of fish (DeLury 1951). A single anode was used and stunned fish were collected with a dipnet. Fish from each site were held in separate plastic tanks with oxygen supply. The total length of each specimen was measured to the nearest cm. Fish of 10 cm or more were individually weighed to the nearest gram. For smaller specimen of less than 10 cm total length, a representative number of at least 15 fish was weighed to determine the condition factor (CF = (weight [g] / total lenght^3 [cm]) x 100) and to determine the total biomass. Individuals of Salmo trutta were classified in the field as wild or stocked according to their distinctive morphological characteristics. In particular, stocked specimens were identified by their distinct size class (all stocked trout had a total length of 35.0 ± 2.0 cm), body and fin shape (fin deformations in the stocked trout) as well as by their distinct colour pattern. All fish were directly released at the sampling sections from which they had been collected. No mortality of the fish was detected.

The fish movement study was carried out from April to June 2009 over 42 days. Fish traps were installed in all fish passes. The fish traps (length = 3.2 m) were made of non-rotting nylon nets, two wings (length = 3 m each), seven opening rings (diameter of the first ring = 0.6 m) and three inlets to hold the caught fish. The meshsize of the wings and of the entrance section of the fish trap was 18 mm, the middle section was 15 mm and the catch bag was 11 mm. Two traps were installed at each site, located at exit and entrance of the fish bypass, blocking the complete channel width. The traps were checked daily at 5 pm and upstream and downstream movements were recorded. The term “downstream movement” refers to both active movements and passive drift in downstream direction. Species were determined and total length (to the nearest 0.5 cm) and weight (to the nearest gram) of each specimen was recorded. After this procedure, all fish were directly released upstream (fish trap for upstream migration) or downstream (fish trap for downstream migration).

Physicochemical habitat characteristics

In order to detect the effects of hydraulic patterns on community composition, the individual flow gradient of each reach was measured with a surveyor’s optical level Topcon AT-G1-3 (Topcon Deutschland GmbH, Willich, Germany). In addition, water depth, current speed 5 cm below surface and 5 cm above stream substrate were measured at each reach comprising ten cross sections, each subdivided into three points at the left, middle and right side of the river (distance to the bank for the left and right measurement points was 0.3 m). River width was measured at each of the ten cross sections. Current speed was measured with a flow measuring instrument HFA (Höntzsch Instrumente, Waiblingen, Germany). Water depth and river width were recorded using a graduated measuring rod with a scale bar in cm.

To detect differences in water chemistry between the sites, dissolved oxygen, temperature, specific conductance and pH were measured in the free flowing water of each site using handheld Multi-340i equipment (WTW, Weilheim, Germany). Generally, a great similarity of these variables among the different sampling sites was observed. Hence these parameters were excluded from further analysis. Only specific conductance was significantly lower in the Riedbach compared to all Nassach sites.

Statistical analysis

Normality of data was tested with the Shapiro-Wilk test and the homogeneity of variances was tested with the Levene-Test. Since all data were not normally distributed, Kruskal-Wallis-ANOVA and – in case of significance ($p < 0.05$) – Mann-Whitney-U post hoc tests with Bonferroni correction were carried out to test for differences between the different sites (1, 2, 3) and reaches (FP, US, DS). In order to evaluate the differences in the fish community structure as well as the interactions of biotic and abiotic stream habitat characteristics, non-metric multidimensional scaling (NMDS) was performed using the R-package vegan (Oksanen et al. 2009). To establish a link between abiotic and biotic variability and fish community composition, Shannon index for diversity, Evenness and species richness were calculated for fishes and abiotic habitat factors. For abiotic habitat factors, the number of different values was treated as species richness for calculation of the Shannon index and Evenness. In addition to diversity values, the mean, minimum and maximum values of all assessed structural habitat variables as well as the flow gradient were considered for the environmental fitting on the NMDS ordination plot (Oksanen et al. 2009). Environmental fitting was performed with 1,000 permutations. Only environmental variables with significant correlation with the NMDS were considered as ordination plot vectors. To detect relations between fish movements and discharge as well as between fish movements and water temperature, linear regressions and Spearman rank correlations were computed. For standardization, the deviation of the daily values of catch and discharge/ water temperature from the median values during the survey period were used for regression analysis. The correlations were performed for a pooled sample of all fishes, as well as for the two mainly migrating species Gobio gobio and Barbatula barbatula. All statistical analyses were carried out using R (version 2.12.0). Significance was accepted at $p < 0.05$.

6.4 Results

Structural habitat characteristics

The structural habitat characteristics measured by water depth, current speed, stream width and flow gradient strongly differed between fish pass (FP) sites and their adjacent sections in the main stream, with differences being most pronounced between FP and upstream sites (US). In the Nassach, the flow gradient was lower in US and DS compared to FP sites with slightly higher values in US compared to DS (Table 6.1). In the Riedbach site the flow gradient of the FP and DS site was similar and had almost the same values as the FP sites of the Nassach. The US site of the Riedbach was similar to the Nassach DS sites (Table 6.1). Water depth and river width did not differ significantly between the two Nassach sites (site 1 and 2) for pooled data over all three reaches (US, DS and FP), while the Riedbach (site 3) generally had lower water depth (Mann-Whitney-U-Test: p (1-3) < 0.001, p (2-3) < 0.001). River width was smaller in Riedbach site 3 than in Nassach site 1 (Mann-Whitney-U-Test: p (1-3) < 0.01). Current speed above stream substrate and at the surface did not differ significantly between sites. The fish passes (FP) generally had significantly lower water depth (Mann-Whitney-U-Test: p (FP-US) < 0.001, p (FP-DS) < 0.001) and smaller river width (Mann-Whitney-U-Test: p (FP-US) < 0.001, p (FP-DS) < 0.001) than the US and DS reaches for pooled data over all sites. In addition, water depth was significantly higher in the US reaches than in the DS reaches (Mann-Whitney-U-Test: p (US-DS) < 0.001), while US and DS reaches showed no difference in river width. Current speed above stream substrate was higher in the fish passes (FP) than in the upstream (US) and downstream (DS) sections (Mann-Whitney-U-Test: p (FP-US) < 0.001, p (FP-DS) < 0.001) and higher in DS than in US sections (Mann-Whitney-U-Test: p (US-DS) < 0.001). For current speed on the surface, exactly the same pattern was detectable (Mann-Whitney-U-Test: p (FP-US) < 0.001, p (FP-DS) < 0.001, p (US-DS) < 0.01). The DS section of the Riedbach (site 3) was not significantly different from the fish passes in terms of surface current speed, current speed above stream substrate and water depth (Mann-Whitney-U-test: $p < 0.05$). Mean values and ranges of structural habitat characteristics are shown in Table 6.1.

SITE	ST	DT [l/s]	FL [%]	LE [m]	DE [cm]	WI [m]	CB [m/s]	CS [m/s]
1	US	300	0.04	210	69.3 [43 - 120]	5.18 [3.8 - 6.7]	0.02 [0 - 0.08]	0.05 [0 - 0.10]
1	DS	300	0.09	210	93.0 [23 - 145]	5.65 [5.2 - 6.5]	0.03 [0 - 0.12]	0.09 [0 - 0.37]
1	FP	300	1.05	210	22.4 [10 - 41]	2.50 [1.5 - 4.0]	0.23 [0.02 - 0.81]	0.35 [0.04 - 0.80]
2	US	300	0.07	210	79.8 [20 - 140]	4.27 [2.6 - 5.8]	0.00 [0 - 0.03]	0.02 [0 - 0.06]
2	DS	300	0.12	210	58.8 [32 - 100]	5.71 [2.8 - 7.8]	0.03 [0 - 0.16]	0.10 [0 - 0.35]
2	FP	300	1.01	210	20.9 [7 - 43]	2.30 [1.8 - 2.7]	0.15 [0.01 - 0.59]	0.28 [0.01 - 0.68]
3	US	100	0.12	330	46.9 [32 - 69]	4.68 [4.5 - 4.8]	0.00 [0 - 0.01]	0.01 [0 - 0.04]
3	DS	100	1.10	330	21.4 [7 - 70]	3.40 [2.6 - 4.2]	0.12 [0 - 0.38]	0.20 [0.03 - 0.56]
3	FP	100	0.95	330	18.4 [5 - 63]	1.83 [1.2 - 2.7]	0.07 [0 - 0.33]	0.14 [0.01 - 0.43]

Table 6.1: Structural habitat characteristics of FP = fish pass, US = upstream site and DS = downstream site. ST = reach, DT = discharge, FL = flow gradient, LE = talweg length of the sampled reach, DE = depth, WI = width, CB = current speed 5 cm above stream substrate, CS = current speed 5 cm below water surface. DE, WI, CB and CS are given as mean values, with minimum and maximum in square brackets below mean values.

Fish community composition

Over all sampling sites in the Nassach and Riedbach, 17 fish species from seven families were found during the electrofishing surveys. In the Nassach, 15 species were detected and in the Riedbach 14 species. The fish assemblage was dominated by cyprinids, comprising eleven species. The most abundant species were Squalius cephalus, Salmo trutta, Barbatula barbatula, Gobio gobio, Leuciscus leuciscus and Rutilus rutilus, but smaller numbers of Pseudorasbora parva, Perca fluviatilis, Carassius gibelio and Tinca tinca were also recorded. Following the habitat classification by Zauner and Eberstaller (1999), only three species were typical rheophilic (Salmo trutta, Barbatula barbatula, and Gobio gobio) whereas the other fish species were described as indifferent or limnophilic. Most common species in the fish passes were wild Salmo trutta, Barbatula barbatula, Gobio gobio and Leuciscus leuciscus. Generally, upstream and downstream sites of the fish passes held more species (US = 17, DS = 13 and FP = 9) but the numbers of specimens were almost balanced between US (CPUE/100 m² = 35.9), DS (CPUE/100 m² = 29.7) and FP (CPUE/100 m² = 33.6) sites. The biomass-related CPUE was five-fold higher in US (CPUE/100 m² = 2.15 kg) and DS (CPUE/100 m² = 2.49 kg) sites compared to FP sites (CPUE/100 m² = 0.44 kg). The locations of the Nassach (site 1 and site 2) and the Riedbach (site 3) differed in the number of fish species, number of individuals and biomass. In the Nassach US and DS locations, species richness and biomass was higher but the number of individuals was lower than in the related fish passes. In the Riedbach, opposite results were observed. Here, the fish pass held four-fold (compared to the US site) and three-fold (compared to the DS site) less individuals than the related US and DS reaches. Generally, the number of individuals caught in the Riedbach (site 3) US (CPUE/100 m^2 = 71.6) and DS (CPUE/100 m^2 = 63.2) was much higher than in the Nassach (CPUE/100 m^2: site 1 US = 8.3 and DS = 8.5; site 2 US = 7.7 and DS = 19.3). In spite of the higher fish numbers in the Riedbach, the biomass of the catch was lower (CPUE/100 m^2: US = 1.2 kg, DS = 0.71 kg and FP = 0.12 kg) compared to the Nassach sites (CPUE/100 m^2: site 1 US = 1.86 kg, .DS = 1.71 kg and FP = 0.54 kg; site 2 US = 4.15 kg, DS = 4.94 kg and FP = 0.74 kg). The three fish passes differed in terms of caught individuals and biomass, with the most productive sites at the Nassach, where site 2 held most fish and the highest biomass (CPUE/100 m^2 = 49.5 individuals/ 0.74 kg) compared to the Nassach site 1 (CPUE/100 m^2 = 37.3 individuals/ 0.54 kg) and the Riedbach (site 3) fish pass (CPUE/100 m^2 = 17.7 individuals/ 0.12 kg). The catch in the fish passes represents a mixture of reophilic and ubiquitous fish species which all could be detected in US and DS sites, too. However, wild Salmo trutta could only be detected in the FP reaches and in the DS of the Riedbach (site 3). Larger ubiquitous fish species like Cyprinus carpio, Esox lucius and Abramis brama were not found in the fish passes. The abundance of rheophilic target species for conservation like wild Salmo trutta or Barbatula barbatula was higher in the fish passes of the river Nassach (site 1 and site 2) than in their related upstream and downstream reaches. In the Riedbach (site 3), the abundance of Barbatula barbatula and wild Salmo trutta was higher in DS than in the respective FP and the respective US location. Also, the rheophilic species Gobio gobio was much more abundant in US (736) compared to FP (13) and DS (116). Smaller fishes and smaller specimen were mostly found at FP sites and the Riedbach US and DS reaches, while bigger fish were exclusively found in US and DS sites (Fig. 6.2). The maximum total fish length in the FP reaches was 34 cm, in the US reaches 80 cm and in the DS reaches 85 cm.

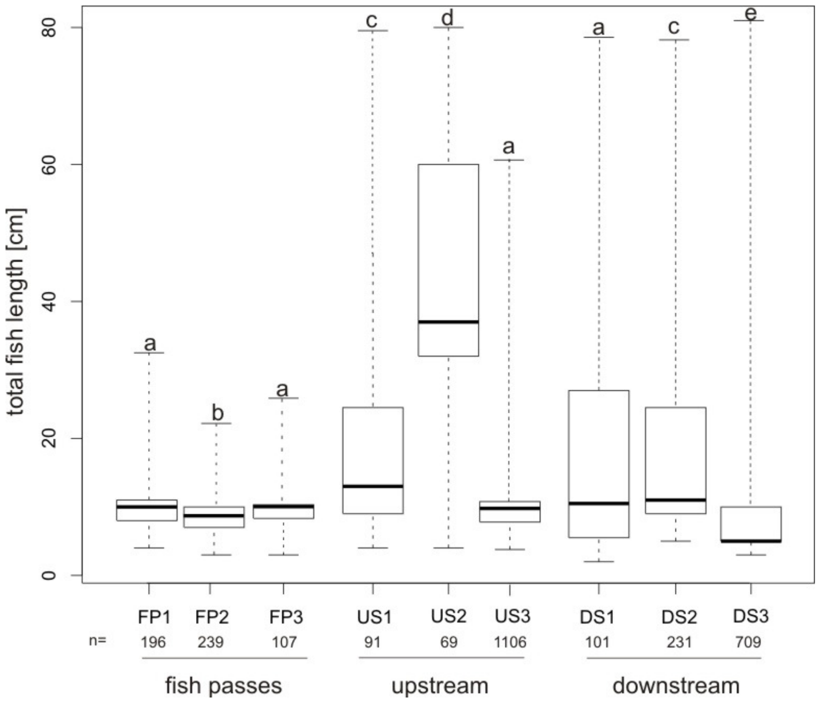


Fig. 6.2: Differences of total fish lengths in the sampled reaches (FP = fish pass, US = upstream site, DS = downstream site) at each of the three sampling locations (1 = Rain-Mill, 2 = Zinken-Mill, 3 = Ried-Mill). Box: 25 % quantile, median, 75 % quantile; whisker: minimum, maximum values. Letters above whiskers (a, b, c, d, e) indicate significant differences ($p < 0.05$, p-values of single comparisons were Bonferroni-adjusted by multiplying them with the number of comparisons), n = number of caught fish.

The results of the NMDS indicate a great similarity of all FP sites with e.g. Barbatula barbatula, wild Salmo trutta, Leuciscus leuciscus and Phoxinus phoxinus as common species. With exception of DS 3, all US and DS reaches form a separate group in the NMDS ordination plot (Fig. 6.3). Characteristic species were e.g. Carassus gibelio, Esox lucius, Squalius cephalus, Rutilus rutilus, Abramis brama, Cyprinus carpio and stocked Salmo trutta. The separation of the fish passes and DS 3 from the other sites is mainly correlated with the mean current speed on the bottom ($r^2 = 0.76$) and surface ($r^2 = 0.79$), the maximum current speed at the bottom ($r^2 = 0.77$) and surface ($r^2 = 0.76$) and the minimum current speed on the surface ($r^2 = 0.61$). The separation between FP and DS 3 to all other DS and US sites was strongly correlated with differences in the flow gradient ($r^2 = 0.88$), the number of values of current speed at the surface ($r^2 = 0.73$), the Shannon index, evenness and number of values of current speed over the bottom ($r^2 = 0.80$, $r^2 = 0.70$, $r^2 = 0.82$), the minimum, maximum and the mean depth ($r^2 = 0.72$, $r^2 = 0.81$, $r^2 = 0.74$), the evenness of river width ($r^2 = 0.71$) and the number of fish individuals ($r^2 = 0.80$) (Fig. 6 3).

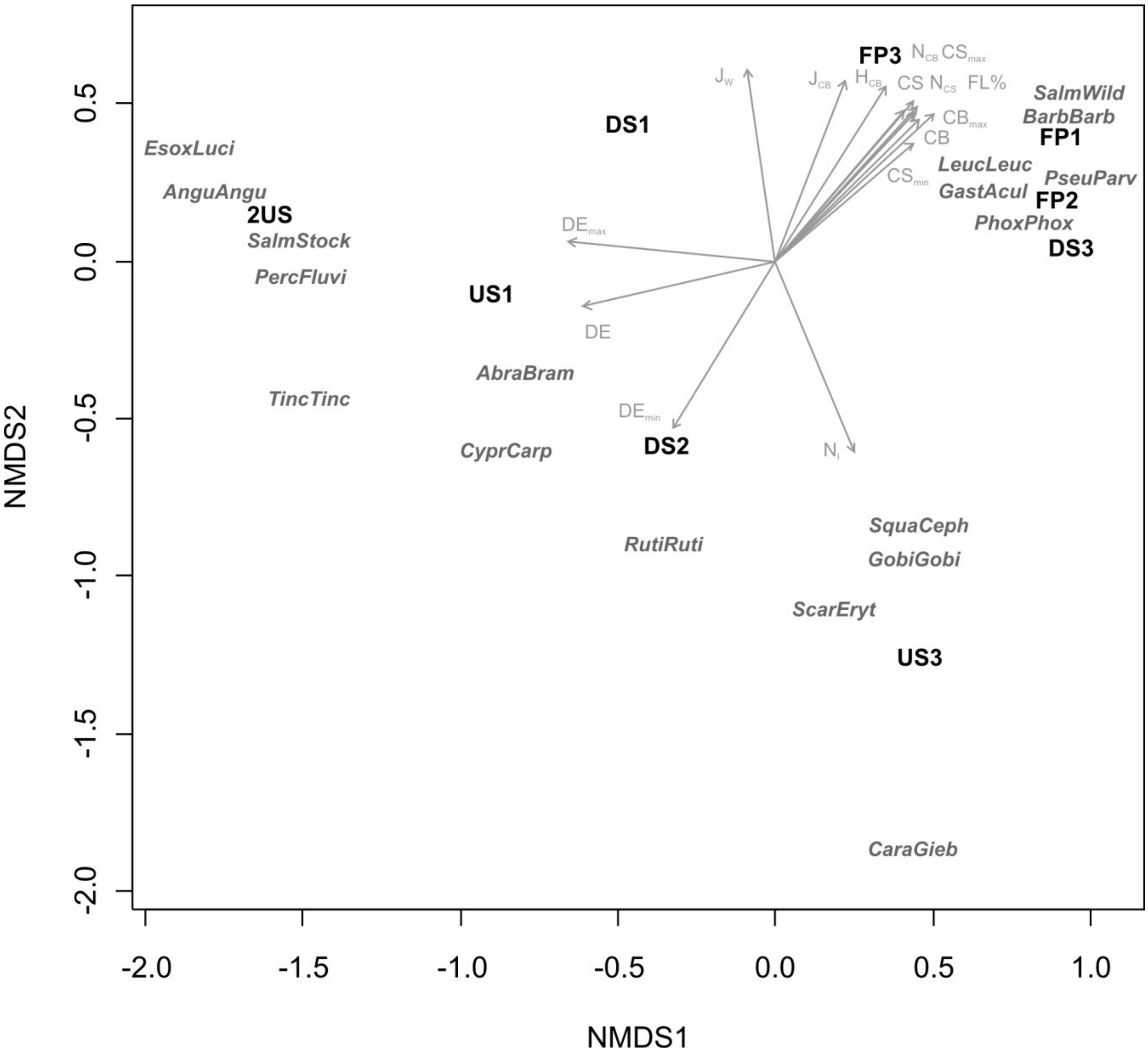


Fig. 6.3: Non-metric multidimensional scaling (NMDS) performed for all three study sites with FP = fish pass, US = upstream site and DS = downstream site. Species are displayed with abbreviations in italics (for full names see table 6.2). Environmental variables ($p < 0.05$ based on 1,000 permutations) are displayed as arrows with DE_{max} = maximum water depth, DE = mean water depth, DE_{min} = minimum water depth, N_I = number of individuals (fish), CS_{min} = minimum current speed below surface, CB = mean current speed above stream substratum, CB_{max} = maximum current speed above stream substratum, FL% = flow gradient, N_{CS} = number of different values for current speed below surface, CS_{max} = maximum current speed below surface, CS = current speed below surface, N_{CB} = number of different values for current speed above stream substratum, H_{CB} = Shannon diversity of current speed above stream substratum, J_{CB} = Evenness of current speed above stream substratum, J_W = Evenness of river width. Stress: 0.026.

Fish movement

Species	Abbreviation	Rain-Mill DS1	Rain-Mill US1	Rain-Mill FP1	Zinken-Mill DS2	Zinken-Mill US2	Zinken-Mill FP2	Ried-Mill DS3	Ried-Mill US3	Ried-Mill FP3
Abramis brama	*AbraBram*	0.4	0.1	0.0	1.4	0.0	0.0	0.0	0.0	0.0
Anguilla anguilla	*AnguAngu*	0.7	1.2	0.0	0.5	2.3	0.0	0.1	0.0	0.0
Barbatula barbatula	*BarbBarb*	3.0	1.2	28.0	0.5	0.1	31.5	36.8	0.7	7.6
Carassius gibelio	*CaraGieb*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0
Cyprinus carpio	*CyprCarp*	0.0	0.3	0.0	0.8	0.0	0.0	0.0	0.1	0.0
Esox lucius	*EsoxLuci*	0.3	0.3	0.0	0.1	0.9	0.0	0.0	0.0	0.0
Gasterosteus aculeatus	*GastAcul*	0.4	0.0	1.5	0.0	0.0	0.0	2.1	1.0	1.7
Gobio gobio	*GobiGobi*	2.2	1.3	4.6	7.2	0.6	9.3	10.3	47.7	2.2
Leuciscus leuciscus	*LeucLeuc*	0.3	0.4	2.1	0.8	0.0	0.2	3.5	0.1	0.5
Perca fluviatilis	*PercFluvi*	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Phoxinus phoxinus	*PhoxPhox*	0.0	0.1	0.0	0.0	0.0	0.2	6.1	1.2	2.3
Pseudorasbora parva	*PseuParv*	0.0	0.0	0.0	0.0	0.0	4.6	0.0	0.3	0.0
Rutilus rutilus	*RutiRuti*	0.4	3.2	0.0	5.7	0.9	0.4	0.6	7.7	0.3
Salmo trutta (stocked)	*SalmStock*	0.3	0.1	0.2	1.3	2.6	0.2	0.0	0.0	0.0
Salmo trutta (wild)	*SalmWild*	0.0	0.0	0.1	0.0	0.0	1.2	0.4	0.0	0.5
Scardinius erythrophthalmus	*ScarEryt*	0.0	0.0	0.0	0.0	0.1	0.0	0.2	0.5	0.0
Squalius cephalus	*SquaCeph*	0.4	0.0	0.0	1.1	0.1	1.9	3.0	11.9	2.6
Tinca tinca	*TincTinc*	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0

Table 6.2: Species list of the fish habitat assessment. DS = downstream site, US = upstream site, FP = fish pass. The catch per unit effort of each species in each site is given in individuals per 100 m².

All three semi-natural fish passes were used by fishes for upstream (UP) and downstream (DO) movements resulting in a total of 438 recorded fish movements. Fifteen fish species and one lamprey used the fish passes as movement corridors, including nearly all of the fish species detected by the electrofishing and an additional two species, *Barbus barbus* and *Lampetra planeri*. Only *Phoxinus phoxinus*, *Abramis brama* and *Pseudorasbora parva* were not caught in the fish pass during the six week survey. The most frequently caught fish were *Gobio gobio* (251 individuals), *Barbatula barbatula* (42 individuals), *Scardinius erythrophthalmus* (28 individuals), *Squalius cephalus* (27 individuals), *Rutilus rutilus* (23 individuals), *Anguilla anguilla* (23 individuals) and *Salmo trutta* (12 individuals), with *Gobio gobio* (detected at 82 % of the survey days) and *Barbatula barbatula* (detected at 46 % of the survey days) being the most consistently caught species. In general, an almost balanced ratio of upstream (235) and downstream (203) movement in the fish passes was detected. At the three sites, all fish species were caught during up- and downstream movements, except for *Barbus barbus*, *Tinca tinca*, *Gasterosteus aculeatus* and *Lampetra planeri* which were only caught once. Most fish movements were detected in site 1 (191, UP 85, DO 106), almost double the number of migrating fish at the Nassach site 2 (104, UP 64, DO 40). In the Riedbach site 3, 143 (UP 86, DO 57) fish were caught in the fish traps. All three sites showed a similar compositions of migrating species with 11 species in site 1 (UP 11, DO 9), 13 species in site 2 (UP 9, DO 11) and 3 (UP 9, DO 10), respectively (Table 6.2).

Analyses of the time-related movement patterns indicated that days with an increased discharge of Nassach and Riedbach correlated with enhanced fish movement, i.e. with more fish species and more individuals using the fish pass (Fig. 6.4). The regression function was mostly determined by the highest discharge events at which the greatest fish movements occurred. The correlation between downstream movements and discharge was smaller than between upstream movements and discharge. This pattern was particularly evident in *Gobio gobio* which increased movement behaviour with a preference on downstream movements (Spearman's rank correlation; US: rho = 0.41, $p < 0.05$, DS: rho = 0.26, $p > 0.05$; linear regression, US: $r^2 = 0.17$, $p < 0.05$, DS: $r^2 = 0.24$, $p < 0.05$). Only a small number of days without fish movements were detected at the three sites, with exception of a period of four days without catch at site 3 due to low water discharge. Water temperature ranged between 13.5 °C and 19.5 °C, but there was no relation between water temperature and fish movements detectable over the survey period ($r^2 = 0.01$, $p > 0.05$). In *Barbatula barbatula*, *Scardinius erythrophthalmus* and *Rutilus rutilus* movement peaked during short time periods. While *Barbatula barbatula* and *Rutilus rutilus* preferred an earlier time period at late April and the beginning of May, *Scardinius erythrophthalmus* preferred the middle of May for enhanced fish movements. At the remaining survey time these fish occurred sporadically in small numbers.

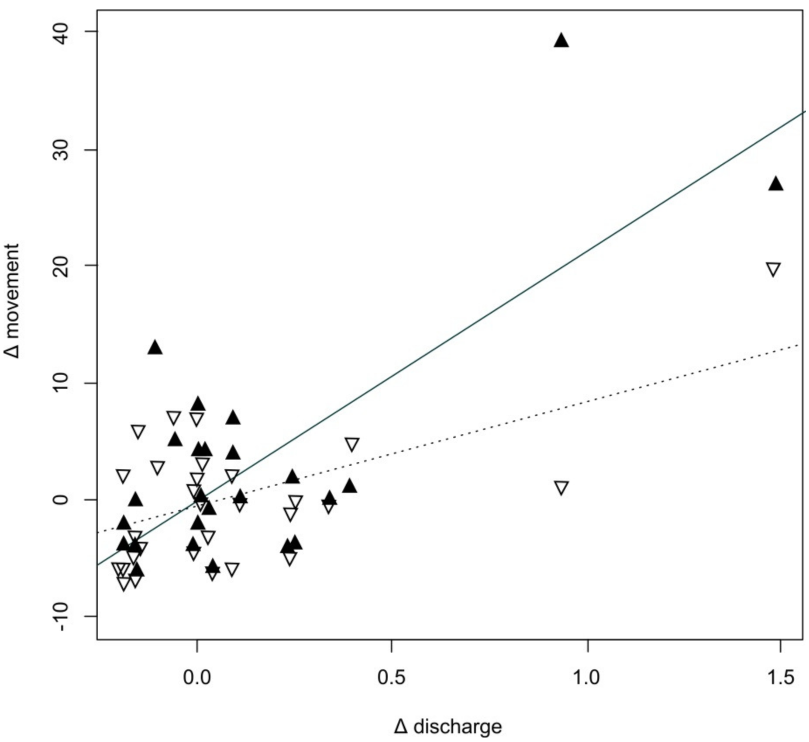


Fig. 6.4: Regression between the deviation of discharge (measured at the water gauge Römershofen in the river Nassach) from average conditions (Δ discharge) with the deviation of upstream (continuous line, upward triangles filled) and downstream (dashed line, downward triangles unfilled) fish movements from average conditions (Δ movements). Upstream: $p < 0.001$, $r^2 = 0.57$, downstream: $p < 0.001$, $r^2 = 0.31$.

6.5 Discussion

Fish habitat use

Free fish movement and a sustainable restoration of degraded fish habitats in heavily modified river reaches are crucial for a successful achievement of the good ecological status, as claimed in the European and German legal acts to the WFD. The implementation of fish passes in practice is often trial and error based, more founded on limited funding than on scientific knowledge (Agostinho et al. 2002, Calles & Greenberg 2009, Kemp & O'Hanley 2010, Pelicice & Agostinho 2008, Roscoe & Hinch 2010). The scientific investigation of fish passes in the past was mostly limited to the functions of these artificial water courses as migration corridors (e.g. Baker & Boubée 2006, Baumgartner and Harris 2007, Bizzotto et al. 2009, Knaepkens et al. 2006, Makrakis et al. 2011). However, the potential suitability of artificial fish passes themselves as compensation habitats for rheophilic fishes is often not considered. Also, weirs can have a strong impact on the adjacent up- and downstream habitats and thus on the overall biological community structure. The novel topic of this study was the comparative assessment of three semi-natural fish passes considering their functions as migration corridors as well as compensation habitats for fish species.

The catch of 17 fish species in the study area resembles the general richness of the fish community in the major tributaries of the River Main. In spite of this, the lack of the former rheophilic fish guild indicates a degradation of the fish community within the study area compared to the historic state (Leuner et al. 2000, Klupp 2000). In the study area, only three (Barbatula barbatula, Gobio gobio and Salmo trutta) out of seven potential rheophilic fish species (Barbatula barbatula, Barbus barbus, Chondrostoma nasus, Cottus gobio, Gobio gobio, Salmo trutta and Thymallus thymallus) naturally inhabiting the tributaries of the River Main (Leuner et al. 2000, W. Silkenat, personal communication) could be detected in the assessed reaches. The catch of nine species and their relative high abundance in the fish passes shows that these artificial flow courses are suitable as habitat for most of the local fish species inventory. Particularly, all rheophilic species detected in this study exclusively occurred in the fish passes in high numbers. Within species, it is remarkable that wild Salmo trutta used the fish passes as habitat while the stocked Salmo trutta were mostly found in adjacent US and DS locations. These differences in the behaviour of stocked versus wild trout indicate that both have different ecological niches and that stocking with hatchery reared trout cannot fully compensate habitat deficites. The occurrence of wild Salmo trutta in the DS of site 3 can likely be explained by a steeper flow gradient than all other US and DS reaches which causes a higher variability and higher absolute values of current speed resulting in higher habitat quality for rheophilic fish species. This is supported by the high abundance of Barbatula barbatula, which also inhabited the fish passes and the DS site 3.

Site 3 had the steepest flow gradient of all DS reaches, due to the steeper valley terrain and its stretched flow course along a small road.

Gobio gobio which is also listed as a rheophilic species used the same habitats like Salmo trutta and Barbatula barbatula, but additionally occurred in very high numbers in the US reach of site 3. This indicates its ability to establish increased populations under limnophilic habitat conditions in heavily modified water bodies (HMWB) (Jurajda 1995, Prenda et al. 1997) and therefore its limited indicator value to detect rheophilic conditions.

The occurrence of mainly smaller fishes in the fish passes supports the assumption that the abundance of microhabitats suitable for small species or juvenile stages of larger species like Salmo trutta, Squalius cephalus or Leuciscus leuciscus is much higher than in the related US and DS reaches. Fish passes with high variability of current speed and water depth seem to be habitats suitable for juvenile fishes (Copp 1992, Jurajda 1999, Humphries et al. 2002). The simultaneous inclusion of abiotic habitat variables and fish assemblage data in the multivariate detrended correspondence analysis support this assumption. In combination with the environmental fitting (Oksanen et al. 2009), the NMDS suggests that the dissimilarity of all FP sites to the Ds and US locations is strongly correlated with channel slope, variability of current speed, water depth and river width. This adequately describes the major structural differences of these sites and is consistent with the results from univariate statistics. The differences between the three fish passes in terms of habitat quality cannot be explained by their abiotic channel structure but are more likely to result from higher discharge of the Nassach sites compared to the Riedbach site. The colonisation of the Riedbach site was partially interrupted during several short time periods of strongly natural fluctuating discharge. This can be explained by a construction-based limited connectivity of the FP 3 to its related US site during minimum discharge extremes.

Although species-specific habitat requirements were not specifically considered during the planning and construction process of the fish passes investigated here, they were clearly found to provide suitable habitat for different life stages of rheophilic species. Slope, variability of current speed, water depth and channel width turned out to be most important of all evaluated variables which determined habitat quality for rheophilic fish species. The fish bypass channels created here might only have met the habitat requirements of these species by chance (e.g. due to the available space and the slope of the area according to which these channels were built) but can serve as examples for successful restoration in other areas where a more thorough planning is required to meet the conservation needs of rheophilic species.

Fish movement

Nearly all of the fish species detected during the electrofishing surveys used the fish passes for upstream and downstream movements, which demonstrates the principle functionality of the bypasses as a compensation for weir-caused barrier effects. The high amount and continuous movement of smaller fish species like Barbatula barbatula and Gobio gobio show that migration behaviour of small fish species can be very intense, whereas current fish bypass channel planning is mostly oriented towards the large fish species (e.g. Gowans et al. 2003, Jensen & Aass 1995, Laffaille et al. 2005, Rivinoja et al. 2001). For this reason, it is highly recommended to consider migration requirements of small fish species in future restoration planning. Fish migrations can be driven by several reasons: dispersal, searching for food, exploration due to changing environmental conditions, or competition and spawning migrations (Lucas & Baras 2001). In this study, several species-specific movement behaviours and time periods of enhanced fish movement could be observed. The movement peak of Barbatula barbatula at the beginning of the fish trap survey can likely be explained by spawning migrations at the end of April and the beginning of May (Vinyoles et al. 2010). This is also true for Rutilus rutilus and later in May for Scardinius erythrophthalmus (Vollestad & L’Abée-Lund 1987, Tarkan 2006). To ensure a high functionality of fish passes, species-specific requirements on flow conditions have to be considered especially during time periods of enhanced migration behaviour. The correlation of movement behaviour to high flow conditions as observed in our study for Gobio gobio, Anguilla anguilla and other species shows that the construction scheme of fish passes should allow dynamically fluctuating flow conditions. A dynamic discharge of the fish pass, following the natural discharge, seems to be a crucial restoration measure to ensure high functionality even during elevated discharge when enhanced fish migrations occur. The three fish passes evaluated in this study are not equally suitable as migration corridors for fishes. The high abundance of fishes in US and DS sites not always resembles the catch efficiency of the fish traps. For example, the US and DS reaches of the Riedbach provided highest abundance of migratory fish species, but the fish trap in the FP showed only average catch numbers for upstream as well as downstream movements in comparison to the Nassach sites. This indicates some restrictions for the passability of the fish pass, partially related to the extreme minimum flow conditions. As there are only marginal differences in flow gradient between the fish passes, a further explanation can be the significantly lower current speeds associated with reduced flows in comparison to the Nassach sites. The Nassach site 2 seems also to have limitations as a fish migration corridor. Since both Nassach sites are similar in construction design, discharge and fish community structure, the marginal differences in the flow gradient from 1.01 % (site 2) to 1.05 % (site 1) may be responsible for the slightly higher current speeds and water depths of site 1 and in conclusion for the reduced fish movements in site 2.

Conclusions and recommendations for fish bypass construction

Migration behaviour of small fish species seems to be widely underestimated in practice. For the future construction of fish bypass channels and conservation management, the requirements of juvenile fish and small fish species should be included in future concepts for stream restoration.

At the same time, the currently underestimated function of fish bypasses as compensation habitats should be considered, since river reaches with high variability of current speed and water depth are widely underrepresented in highly modified water bodies. They can be important key habitats for the reproduction of rheophilic species, as well as for juveniles and small fish species. Consequently, well-designed fish bypass channels can have positive effects on the overall fish diversity in streams affected by weirs by enriching overall habitat diversity.

The construction of fish passes needs to address the required reliability during peak flows, as well as a wide range of flow conditions throughout the year. To enable fish species to respond to flood events with increased migration behaviour, the flow conditions in the water course should follow the natural dynamics of the discharge. In this context, it is important to design fish bypass channels in a way that they do not lose functionality due to hydraulic overstressing or low flow conditions. The individual flow gradient and its related structural variables of the artificial water course emerged to be the most important factors determining fish community structure and distribution, as well as the functionality as migration corridor.

The combination of channel slope (0.95% - 1.05%), river width (1.2 m – 4.0 m), water depth (5 cm – 63 cm) and discharge (100 l/s - 300 l/s) of the fish passes investigated in this study indicates that they provide suitable habitat for rheophilic species and can thus be assessed as improvements of overall habitat quality. However, low slopes generally increase the length and cost of fish pass channels, particularly when adjacent land is scarce and expensive. Consequently, the construction of these fish bypass channels can be unaffordable for medium high obstacles. Under these circumstances, a combination of technical fish passes and nature-oriented bypass channels are often the only option to improve habitats for rheophilic species.

As evident from the study presented here, fish bypass channels can make major contributions to fish habitat improvements, but ultimately, the restoration of the river continuum must not be reduced to restoring small habitat sections and migrations of selected fish species, but also needs to consider overall biodiversity and ecological functioning of river systems.

7 The effects of weirs on structural stream habitat and biological communities

M. Mueller, J. Pander & J. Geist
Journal of Applied Ecology 2011, **48,** 1450-1461

7.1 Summary

1. Most of the world's rivers are affected by dams and weirs. Information on the quantitative and qualitative effects of weirs across biological communities is crucial for successful management and restoration of stream ecosystems. Yet, there is a lack of comprehensive studies that have analysed the serial discontinuity in direct proximity of weirs including diverse taxonomic groups from algae to fish.

2. This study compared the abiotic stream habitat characteristics upstream and downstream of weirs as well as their effects on the community structure of periphyton, aquatic macrophytes, macroinvertebrates and fish at five different study rivers.

3. Physicochemical habitat characteristics discriminated strongly between upstream and downstream sides of weirs in terms of water depth, current speed, substratum composition and the transition between free-flowing water and interstitial zone. Accordingly, abundance, diversity, community structure and functional ecological traits of all major taxonomic groups were indicative of serial discontinuity, but the discriminative power of individual taxonomic groups strongly differed among rivers.

4. The simultaneous inclusion of abiotic habitat variables, taxonomic diversity and biological traits in multivariate non-metric multidimensional scaling (NMDS) was most comprehensive and powerful for the quantification of weir effects. In some cases, the intra-stream discrimination induced by weirs exceeded the variation between geographically distant rivers of different geological origin and drainage systems. Community effects were generally detectable on high levels of taxonomic resolution such as family or order level.

5. *Synthesis and applications.* River sections in spatial proximity to weirs are affected seriously and should be included in the ecological assessments of the European Water Framework Directive. Multivariate models which include several taxonomic groups and physicochemical habitat variables provide a universally applicable tool for the ecological assessment of impacts on serial discontinuity and other stressors on stream ecosystem health.

7.2 Introduction

The introduction of weirs into rivers is considered a major threat to aquatic biodiversity (Bunn & Arthington 2002). Alterations of hydraulic components can change the availability of habitat space, habitat quality and the structure of aquatic communities (Brunke et al. 2001, Almeida et al. 2009). The “Serial Discontinuity Concept” (Ward & Stanford 1983) describes the effects of physical barriers such as weirs and dams on biotic and abiotic components of lotic systems in a hypothetical framework, but experimental studies into the ecological effects of weirs have mostly focused on single rivers and single taxonomic groups. Habit et al. (2007) and Santos et al. (2006) could detect changes in the fish community at upstream sides of hydropower plants, Zhou *et al*. (2008) showed effects of a small dam on riverine zooplankton composition and Bredenhand and Samways (2009) recorded a serious decline in macroinvertebrate diversity below a dam in a small river. For a comprehensive assessment of the weir-induced serial discontinuity, it is essential to compare upstream and downstream sides of weirs in their abiotic and biotic habitat characteristics including all major taxonomic groups. This is important since there is recent evidence that cross-taxon congruence in diversity and community composition of aquatic organisms is typically low (Heino 2010). Consequently, comprehensive studies which assess the effects of human impacts on stream ecosystem health (i.e. on aquatic habitat quality and multiple biotic assemblages) are urgently needed.

The main objective of the study presented here was to analyse how abiotic stream habitat characteristics and biotic community effects in the taxonomic groups of periphyton, macrophytes, macroinvertebrates and fishes differ among upstream and downstream sides of weirs, located within carbonate and silicate streams in the three major drainage systems Elbe, Main/Rhine and Danube. Specifically, we hypothesize that upstream and downstream sides of weirs within one river differ in abiotic habitat characteristics, biodiversity and community composition and test if different taxonomic groups (of different trophic levels) differ in their response to weirs. Furthermore, we hypothesize that multivariate methods which include abiotic (water depth, current speed, substratum composition, water chemistry) and biotic effects on different taxonomic levels (community composition, abundance, biomass, functional groups) are more suitable for the quantification of weir effects compared to the univariate consideration of single taxonomic groups.

7.3 Material and methods

Study area

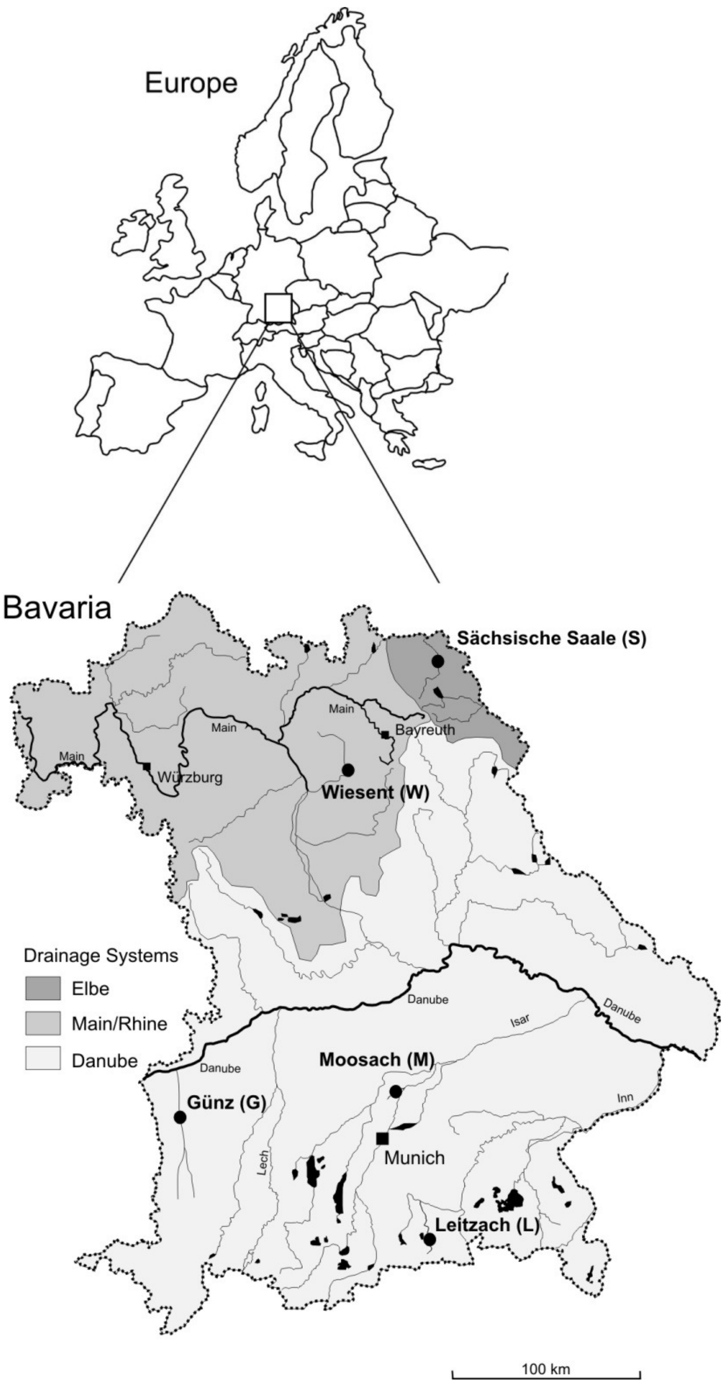


Fig. 7.1: Location and map of the study area. The three major drainage systems (Elbe, Main/Rhine, Danube) are shown in different shades of grey. Study rivers are marked with black dots. For study river details see Table 7.1.

The study was carried out between May and July 2009 at five different study rivers distributed throughout major geological units in Bavaria, Germany (Fig. 7.1). All rivers are located in an area of similar climatic conditions and have similar flow regimes (Table 7.1) which are governed by snow melt-induced peak flows during spring. All rivers were altered by weirs for hydroelectric power production, which form barriers for fish migration. In this study, the term weir refers to a style of dam which is routinely overtopped by water. The sections above dams (referred to as upstream sides, U) reveal strongly altered velocity distributions while downstream sides (D) more resemble the natural flow. None of the study stream sections is affected by hydropeaking regimes. In each river, U and D sides were compared using a standardized sampling design comprising 15 replicates in each side (Fig. 7.2). The length of the sampled river sections on each side was adjusted to the fifteen-fold river width at respective weir sides, resulting in study sections of 150 m to 420 m (Fig. 7.2). For safety reasons, the area in direct proximity of the weirs (two-fold stream width distance) was excluded. This study was designed to evaluate the effects of serial discontinuity in direct proximity of weirs, since the underlying effectors would be disguised with increasing distance (Ward & Stanford 1983). We are aware that the effects of weirs can exceed those observed in the study area.

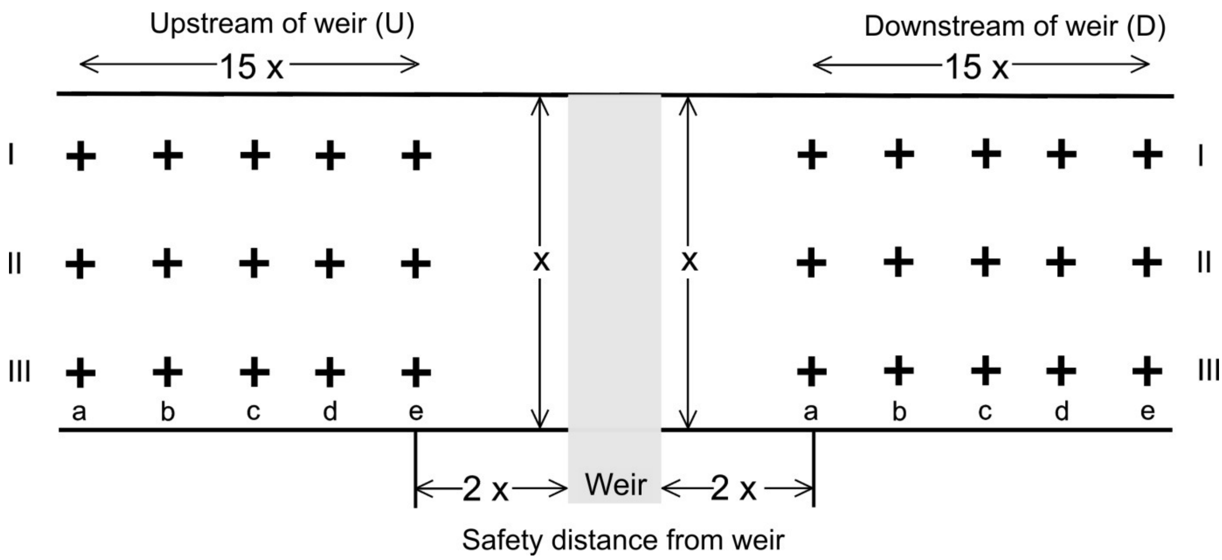


Fig. 7.2: Schematic of the sampling design with: I – III = tracks (for the fish sampling), a – e = labelling for sampling points of each track, + = sampling points (for the sampling of physicochemical habitat variables, periphyton, macroinvertebrates and macrophytes), x = average width of the river measured 50m upstream (U) and downstream (D) of the weir, 15x = length of the sampling area, 2x = area excluded from the assessment for safety reasons.

		Günz (G)	Leitzach (L)	Moosach (M)	Sächsische Saale (S)	Wiesent (W)
Catchment area	[km^2]	526	112	175	523	432
Drainage		Danube	Danube	Danube	Elbe	Main
Geology		Molasse	Limestone alps	Moraine	Basement	Chalkstone
Mean annual discharge	[m^3s^{-1}]	8.35	4.65	2.64	5.41	7.48
Year of construction		1945	1899	17th. c.	a 1905	1924
Heigth of weir	[m]	4.0	4.2	1.3	1.5	1.8
Average river width	[m]	24	14	20	14	20
pH value		7.8	8.0	7.7	7.2	7.9
Specific conductance	[µS/cm]	556	470	762	292	635
Dissolved oxygen	[mg/l]	9.3	10.9	9.6	8.3	10.0
Temperature	[℃]	18.5	9.6	13.5	17.8	14.3
Redox potential	[mV]	580	470	520	490	460

Table 7.1: Characterization of the five study streams: Catchment characteristics, geology, discharge, weir construction details, water chemistry (mean values from field sampling dates).

Physicochemical habitat characteristics

Since substratum characteristics exert significant control on the quality of streambed habitat and benthic community structure (e.g. Geist & Auerswald 2007), the composition of the stream substratum was investigated at 15 points in each U and D site (Fig. 7.2). Substratum was sampled with a steel corer of 8 cm diameter (riverside corer, Eijkelkamp Agrisearch Equipment, Giesbeek, Netherlands). Grain sizes were fractioned with a wet-sieving tower

(Fritsch, Idar-Oberstein, Germany) of decreasing mesh sizes (63 mm, 20 mm, 6.3 mm, 2.0 mm and 0.85 mm). The fractions retained on each sieve were dried at 100 °C and weighed to the nearest gram. The percentage of each grain fraction was determined and the geometric mean particle diameter (*dg*) was calculated according to Sinowski and Auerswald (1999). For a hydraulic characterization, water depth, current speed below surface and 15 cm above ground were measured at each sampling point using a HFA flow-measuring-instrument (Höntzsch Instrumente, Waiblingen, Germany). Dissolved oxygen, temperature, specific conductance, redox potential and pH were measured in the hyporheic zone in 10 cm depth and in the free-flowing water. Water extractions from the hyporheic zone and redox potential measurements were carried out as described in Geist and Auerswald (2007). Dissolved oxygen, temperature, specific conductance and pH were measured using handheld Multi-340i equipment (WTW, Weilheim, Germany).

Periphyton

As most of primary production in medium-sized streams is related to the algal biomass associated with epilithal biofilms (Müllner & Schagerl 2003), periphyton can play an important role for the assessment of the functionality of stream ecosystems. At each sampling point, periphyton was scraped off a 1-4 cm^2 total surface area from all available substratum types (stones and dead wood) using a kitchen knife and a flexible plastic tablet to determine surface area. The sampled periphyton mass was dissolved in 200 ml of water and preserved with 20 ml of acidified Lugol's iodine solution (80% Lugol's iodine solution, 10% glacial acetic acid, 10% methanol). The Utermöhl technique (Utermöhl 1931, DIN EN 15204 2006) was applied before cell numbers were counted using an inverted microscope. Periphyton samples were left to settle for at least 24 h and the sample volume for sedimentation was adjusted to 1-50 ml depending on particle concentration in the sample. Algae were determined according to Ettl et al. (1978-1999) and Cox (1996).

Macrophytes

At each sampling point, all aquatic macrophytes and macroalgae were collected from a surface area of approximately 20 m^2 using a garden rake according to the methodology described in Deppe and Lathrop (1993). Sampling was continued until no additional species was found (typically ~15 min). Species were determined according to Weyer and Schmidt (2007). Macroalgae were determined to genus level according to John et al. (2002). The dominance of macrophyte species was calculated as percentage of sampling points at which the particular species was present.

River	Side	Depth [m]		v a [m/s]		v b [m/s]		dg [mm]		Δ O2 [mg/l]		Δ T [°C]		Δ pH		Δ Eh [mV]		Δ sc [µS/cm]	
G	U	**2.22**	0.27	**0.11**	0.03	**0.19**	0.06	25	7	4.6	1.8	-1.0	0.7	0.2	0.2	95	36	16	22
	D	**1.44**	0.23	**0.29**	0.09	**0.42**	0.08	24	7	4.5	1.4	-0.7	0.4	0.3	0.2	147	93	6	75
L	U	**0.93**	0.21	0.43	0.20	**0.73**	0.28	45	16	5.6	2.7	-0.8	0.3	0.2	0.2	54	41	-39	51
	D	**0.31**	0.14	0.40	0.47	**0.45**	0.52	24	6	5.8	2.0	-0.8	0.4	0.3	0.1	67	46	-33	71
M	U	**1.42**	0.17	**0.11**	0.05	**0.22**	0.05	3	1	7.1	2.4	-2.7	1.1	0.3	0.2	127	72	-13	78
	D	**1.04**	0.33	**0.20**	0.14	**0.32**	0.15	8	2	6.1	2.4	-2.3	1.3	0.6	0.3	82	53	-27	72
S	U	**1.57**	0.35	**0.02**	0.01	**0.05**	0.02	**9**	2	4.4	1.9	**-0.8**	0.5	0.2	0.2	195	24	-172	297
	D	**0.67**	0.25	**0.13**	0.05	**0.20**	0.09	**30**	8	4.1	1.7	**-0.1**	0.3	0.2	0.1	160	55	-289	317
W	U	**1.58**	0.44	**0.07**	0.08	**0.20**	0.11	**10**	3	4.8	2.1	-1.1	0.5	0.4	0.3	**135**	32	**-39**	69
	D	**0.69**	0.45	**0.19**	0.14	**0.40**	0.21	**50**	19	2.4	2.1	-1.1	0.6	0.4	0.3	**66**	47	**-16**	25
Pooled	U	**1.55**	0.51	**0.15**	0.17	**0.28**	0.27	**6**	9	5.3	2.3	-1.3	1.0	**0.2**	0.2	117	65	-49	153
	D	**0.83**	0.48	**0.24**	0.24	**0.36**	0.27	**11**	12	4.5	2.3	-1.0	1.0	**0.3**	0.2	105	72	-65	188
% significant differences		100		80		100		60		20		20		20		20		20	

Table 7.2: Physicochemical habitat characteristics: v a = current speed 15 cm above ground, v b = current speed 10 cm below water surface, dg = geometric mean particle diameter, Eh = redox potential, sc = specific conductance, Δ = difference between the free-flowing water and interstitial zone; small numbers next to values indicate SD; bold numbers show significant differences between respective U and D sides.

Macroinvertebrates

Macroinvertebrate samples were collected with a Surber sampler (Surber 1930) at 15 sampling points for each U and D side. The substratum inside the sampling area of 0.096 m^2 at each sampling point was vigorously disturbed for two minutes to a depth of 10 cm using a metal fork. Makrozoobenthos was then collected in the net (mesh size 0.25 mm) and preserved in 30% ethanol. Macroinvertebrates were classified according to Nagel et al. (1989) using a binocular microscope. Classification was performed on species, family (Chironomids, some Trichoptera and Ephemeroptera) or order level (few Diptera).

Fishes

Fish sampling was conducted using a boat-based electrofishing generator (EL 65 II, Grassel, Schoenau, Germany). Each D and U side was subdivided into three separate tracks (I-III in Fig. 7.2) which were sampled from downstream to upstream direction by the same electrofishing crew. A single anode was used and stunned fish were collected with a dipnet. Fish of each track were kept in separate plastic tanks with oxygen supply. The total length of all specimens was measured to the nearest 0.5 cm. Fish of 10 cm or more were individually weighed to the nearest gram. For smaller specimens, a representative number of at least 15 fish was weighed to determine the condition factor and to determine the total biomass as described in Pander & Geist (2010a). After sampling all three tracks, the fish were released.

Univariate data analysis

In order to assess the exchange between the free-flowing water and interstitial zone, the difference of dissolved oxygen concentration, temperature, pH and redox potential was calculated per sampling point. The catch per unit effort (CPUE) of fish (abundance per 100 m^3, fish biomass in g/100 m^3), macroinvertebrate abundance, number of periphyton cells per cm^2 and species richness of each taxonomic group per sampling point (for all groups except fishes) / track (fishes) and arithmetic means for each U and D side for each river were calculated. Normality of data was tested with the Shapiro-Wilk test and the homogeneity of variances was tested with the Levene-Test. Since data were not normally distributed, Mann-Whitney-U tests were performed for comparisons between pooled U and D sides over all rivers. For multiple comparisons between sides and rivers, Kruskal-Wallis-ANOVA and – in case of significance – Mann-Whitney-U post hoc tests were carried out. Bonferroni correction was applied for multiple testings. All statistical analyses were performed in the open source software R (R Development Core Team 2008).

Shannon index (H), maximum diversity (Hmax) and evenness (E) were computed for all taxonomic groups at each U and D side using the R-package vegan (Oksanen *et al*. 2009). The saprobic index (SI) (DIN 38410-1 2004), the FRI (Index of Fish Regions, Dußling et al. 2005) and the following ecological traits were determined for pooled data of all sampling points/tracks per side: Fishes were assigned to categories of flow current preference, feeding type and structural requirements (Jungwirth et al. 2003). Macroinvertebrates were assigned to functional feeding groups (FFG) (Merrit & Cummins 1996), locomotion types (Moog 1995) and flow current preference (Schmedtje & Colling 1996). The FFGs “filtering collectors” and “gathering collectors” were grouped as “collectors” and the locomotion types “swimming/diving”, “borrowing/boring” and “sprawling/walking” were grouped as “active moving”. The percentage of individuals from each functional trait was calculated per study side and additionally compared over pooled U and D sides from all study rivers. Additionally, the percentage of Ephemeroptera, Plecoptera and Trichoptera (EPT%) was calculated for each side. Characteristic indicator species for U and D sides were determined using one-way SIMPER analysis in Primer v. 6 (Plymouth Marine Laboratory, Plymouth, United Kingdom). Untransformed species count data of all taxonomic groups, pooled for each U and D side, was used as input data, with Bray Curtis similarity for the resemblance matrix and a cut off value for low contributions of 90%.

Multivariate data analysis

In order to detect differences in the response of different taxonomic groups, non-metric multidimensional scaling (NMDS) was performed using taxa abundance data of each of the four groups as input variables for the function metaMDS of the R-package vegan (Oksanen et al. 2009). For a comprehensive assessment, NMDS was performed with the input matrix containing physicochemical habitat characteristics and functional traits of each taxonomic group. The resemblance matrix was calculated in Primer v.6 based on Euclidian distances of the sampling sides for the variables FRI, saprobic index, FFGs, EPT%, percentage of active moving taxa, percentage of rheophilic macroinvertebrates, species richness (for all taxa), eveness (for all taxa), CPUE of fish and macroinvertebrate abundance, cell number of periphyton, abundance of macrophytes, fish biomass, water depth, and current speed below surface. For homogenizing different measurement scales before calculating the distance matrix, raw data were normalized using the pre-treatment normalization function in Primer v.6. For a validation of this NMDS method, regular NMDS and detrended correspondence analysis (DCA) based on commonly used taxa abundance data of all taxonomic groups were performed using functions metaMDS and decorana of the R-package vegan (Oksanen et al. 2009). In order to test the discrimination of U and D sides at different levels of taxonomic resolution, NMDS and DCA was compared for all taxa on the species, family and order level. Environmental fitting on all NMDS plots was performed with 1,000 permutations. Only environmental variables with significant ($P \leq 0.05$) correlation with the NMDS were considered as ordination plot vectors. In addition, β-diversity was calculated as species turn-over (β_t) between U and D in each river for fishes, macroinvertebrates, aquatic macrophytes, periphyton and for all taxa with the function betadiver (R-package vegan) using index g (Koleff et al. 2003).

7.4 Results

Physicochemical habitat characteristics

Physicochemical habitat characteristics discriminated strongly between upstream (U) and downstream (D) sides of the weirs (Table 7.2). Water depth was significantly higher (mean depth U=1.55 m, D=0.83 m, $P<0.01$) and current speed below surface and above ground was significantly lower (mean v a U=0.15 ms^{-1}, D=0.24 ms^{-1}, mean v b U=0.28 ms^{-1}, D=0.36 ms^{-1}, $P<0.01$, respectively) in U than in D sides. Substratum composition differed significantly between U and D sides as measured by geometric mean particle diameter (*dg*), percentage of fines and the fraction >63 mm. Mean particle size *dg* in D was nearly twice the value of U ($P<0.05$). The percentage of fines in D was 9% lower than in U (mean D=28%; mean U=37%, $P=0.029$) and the fraction >63 mm was 7% higher in D compared to U (mean D=10%, mean U=3%, $P<0.05$). The differences in substratum composition were also reflected in the water chemical gradients between free-flowing water and interstitial zone. For instance, differences in the concentrations of dissolved oxygen between free-flowing water and the interstitial zone were 20% higher in U than in D ($P<0.05$). Similarly, gradients in temperature (0.3 °C higher in U than in D, $P<0.05$), and in pH (0.1 higher in D than in U, $P<0.05$) were observed. Differences in redox potential and specific conductance were least discriminative between U and D due to high standard deviations (Table 7.2).

Periphyton

A total number of 129 periphyton taxa was identified. Species richness was significantly lower in the river S than in all other study rivers ($P<0.01$) and cell numbers per cm^2 differed significantly between the five study rivers ($P<0.01$, Fig. 7.3). Over all study rivers a consistent trend towards higher species richness and cell numbers in D sides could be observed (Fig. 7.3), with two additional periphyton species in D compared to U (mean U=10; mean D=12, $P<0.05$), and the number of cells per cm^2 being 40% lower in U than in D (mean U=611,406; mean D=993,605, $P<0.01$). Significant differences in cell numbers per cm^2 between the U and the D side of single study rivers were found in the rivers G and S, with 17-fold higher cell counts in the D than in the U side of river G ($P=0.05$) and 2-fold higher cell counts in the D than in the U side of river S ($P<0.01$) (Fig. 7.3). Beta diversity between U and D was very similar between study rivers, ranging from 0.22 to 0.33 (Table 7.5). Characteristic periphyton taxa according to SIMPER analysis were Chlorophyceae and Cyanophyceae for U sides and Diatoms from the genera *Navicula* and *Gomphonema* for D sides.

Macrophytes

Species richness of aquatic macrophytes was generally low and strongly variable among study rivers. Overall, 18 species of macrophytes from 13 families were found. Numbers of species were almost balanced in U and D (total number of species U:15; D:16) with a slightly higher mean species richness in D (species richness U: mean=5; D: mean=6, Fig. 7.4). Species richness, Shannon index and evenness were not significantly different between U and D. Macrophyte dominance was higher in U sides of three rivers in comparison to the corresponding D sides (U-G 67%, D-G 53%; U-L 100%, D-L 93%; U-M 100%, D-M 80%), lower in U-S (20%) than in D-S (40%) and equal in U-W and D-W (100%). The Shannon index of macrophytes was higher in D than in U sides by a factor of 1.4 (mean U=0.87; mean D=1.18). Beta diversity values also indicated great variability among rivers, with the greatest differences between U and D found in the river S and the lowest difference in the river G (Table 7.4). Only one characteristic species for D, *Fontinalis antipyretica* HEDW., and no characteristic species for U was identified by SIMPER analysis.

Macroinvertebrates

A total of 11,921 specimens from 93 species of macroinvertebrates comprising 51 families were identified. The most common taxa were Diptera (23%), Amphipoda (16%), Ephemeroptera (15%), Plecoptera (10%), Coleoptera (5%) and Trichoptera (4%). The most abundant functional feeding groups were collectors (52%) and shredders (36%), whereas predators (2%), scrapers (2%) and all other groups (8%) were less abundant. Macroinvertebrate abundance was significantly lower in the river G than in all other study rivers ($P<0.01$, Fig. 7.3). Differences in abundance between the U and D within one study river were most pronounced for the rivers W, S and G ($P<0.01$, Fig. 7.3). Species richness differed significantly among study rivers ($P<0.05$) except W-L and S-M. Significant differences in species richness between U and the D were found in the rivers S and W. Pronounced differences between U and D (U:64 species, mean=18.2, and D:81 species, mean=25.2, $P<0.01$) occurred, whereas Shannon index and evenness were less discriminative (except for river S with a 2.3 times higher Shannon index and a 1.6 times higher evenness in D-S than in U-S, Fig. 7.4). Beta diversity as a measure of similarity indicated strong differences in community composition between U and D (Table 7.4). EPT%-values were up to four-fold higher in D sides, with differences varying strongly between streams (Table 7.3). Characteristic macroinvertebrate taxa according to SIMPER analysis were Oligochaeta and Chironomidae for U and the rheophilic taxa *Leuctra nigra* (Plecoptera) and *Rhyacophila* spp. (Trichoptera) for D.

The observed differences in abundance, species richness and diversity were even more pronounced considering the functional traits of these groups. Concerning the flow current preference, differences in the percentage of rheophilic taxa of up to 64% between respective

D and U sides were observed, with a trend towards higher abundance of rheophilic taxa in D compared to U in three of the streams (Table 7.3). Accordingly, the percentage of active moving taxa was lower in U sides than in D sides for all study rivers (U:32%, D:49%). A classification according to functional feeding groups indicated greater abundance of collectors in U sides (U:59%, D:49%) and of shredders in D sides (U:23%, D:41%). The saprobic index was higher in U sides compared to the respective D sides (except for L, Table 7.3).

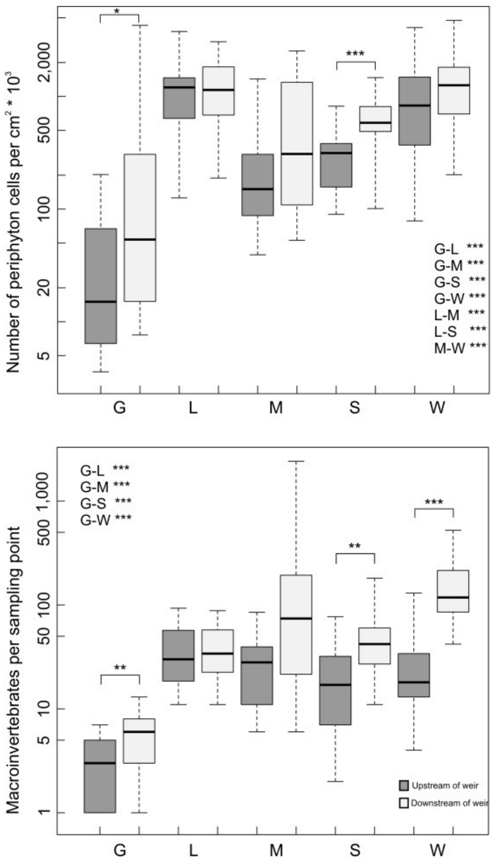


Fig. 7.3: Characterization of periphyton and macroinvertebrate abundance in U and D sides (15 replicates each) of the five study streams: G, L, M, S and W refer to the different study streams, as described in Table 7.1. Box: 25% quantile, median, 75% quantile; whisker: minimum, maximum values. Square brackets between boxes show significant differences in single comparisons within one study river. Significant differences between study rivers are given as text. Significance levels are indicated as follows: 0.01<*P* ≤0.05*, 0.001< *P* ≤0.01**, *P* ≤0.001***.

Fishes

Overall, 27 species from 9 families and one lamprey species (*Lampetra planeri*) were sampled, comprising a total of 2,508 specimens and a total biomass of 244 kg. Species richness was higher in D than in U over all study rivers (U:19 species, mean=7.4, D:23 species, mean=9.6, Fig. 7.4). The CPUE per number of specimens was significantly higher in D than in U (mean U=4.9 per 100 m^3, mean D=5.8 per 100 m^3, P<0.05). The CPUE biomass was three times higher in D than in U (mean U=270 g/100 m^3, mean D = 851 g/100 m^3, P=0.01). Fish diversity was higher and more even in D than in U (Shannon D:2.37, evenness D:0.74, Shannon U:2.05, evenness U:0.68, Fig. 7.4). Species richness was most discriminative between U-S (8) and D-S (13) and between U-G (10) and D-G (14). In contrast to the other study rivers, there were two more species in U-L (5) than in D-L (3). Fish diversity was higher in D-G, D-M, D-S and D-W than in the corresponding U sides and more even in D-W, D-G and D-L than in the corresponding U sides. Beta diversity values ranging from 0.40 to 0.64 indicated pronounced differences in species composition between U and D (Table 7.4). In addition to the differences in abundance, species richness, diversity and community composition among D and U, the highly different fish regions index (FRI) between U (5.89) and D (4.91) sides (stream-specific difference between U and D of 0.02-2.15)

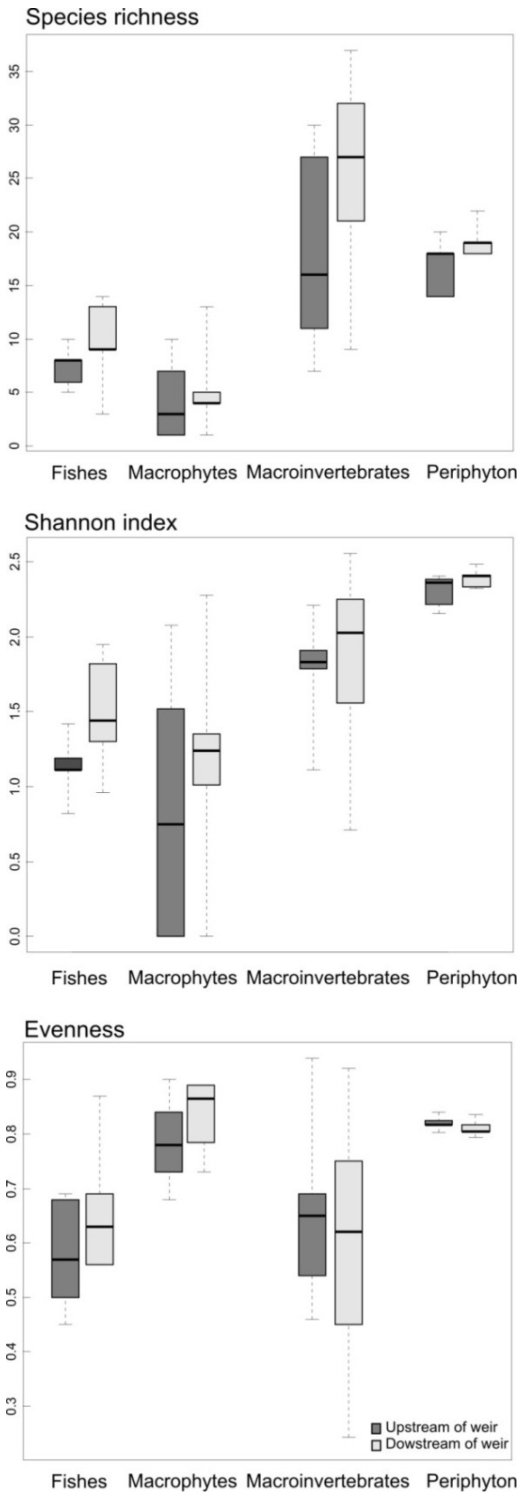


Fig. 7.4: Comparison of species richness and diversity indices of the investigated taxonomic groups in U and D sides: data are pooled for U and D sides in each study river, resulting in 5 replicates each box except for Shannon index and Evenness of macrophytes (4 replicates). Box: 25% quantile, median, 75% quantile; whisker: minimum, maximum values.

indicated pronounced weir effects on fish community structure and the availability of ecological niches for rheophilic specialists (Table 7.3). The difference in the FRI mostly results from the higher abundance of rheophilic species such as *Salmo trutta* L., *Thymallus thymallus* L., *Cottus gobio* L., *Gobio gobio* L., *Barbatula barbatula* L. and *Barbus barbus* L. in D (59%) than in U (24%). The most characteristic fish species according to SIMPER analysis were *Rutilus rutilus* L. for U sides and *S. trutta*, *C. gobio*, *G. gobio* and *Squalius cephalus* L. for D sides. In addition to flow current preference, the fish community composition of U and D also represented different feeding types and structural requirements, with lower abundance of benthivoric and habitat structure-specialised species in U than in D sides in all rivers except the river W (Table 7.3).

		Macroinvertebrates						Fishes			
River	Side	SI	Shredders [%]	Collectors [%]	Rheophilic [%]	EPT%	Active moving [%]	FRI	High structural requirments [%]	Benthivor [%]	Rheophilic [%]
	U	2.20	2	73	33	7	5	6.39	25	0	0
G	D	2.08	4	72	10	26	18	6.09	39	29	29
	U	1.54	27	62	72	45	63	4.06	69	71	71
L	D	1.63	40	52	72	48	72	4.05	77	77	77
	U	1.87	32	34	22	6	42	6.56	31	4	4
M	D	1.85	91	7	86	7	96	4.41	79	57	57
	U	2.11	16	45	0	19	18	6.96	4	0	0
S	D	1.91	3	65	3	34	19	6.10	11	11	11
	U	1.91	13	70	43	37	34	3.96	87	98	96
W	D	1.77	13	76	66	38	39	4.16	80	96	95
	U	1.96	23	59	34	23	32	5.89	43	35	24
pooled	D	1.90	41	49	47	31	49	4.91	57	54	59

Table 7.3: Ecological traits of macroinvertebrates and fishes; values refer to the percent ratio of the number of individuals in relation to all specimens.

Multivariate data analysis

The consideration of single taxonomic groups instead of comprehensive community response analysis in NMDS revealed strong river-specific patterns (Fig. 7.5, Table 7.4). For instance, there was a strong separation between U and D in the river M for fishes and macroinvertebrates, but weak to no separation for periphyton and macrophytes, respectively. In contrast, differences in the river S were greatest for periphyton and macrophytes, but less pronounced for macroinvertebrates and fishes.

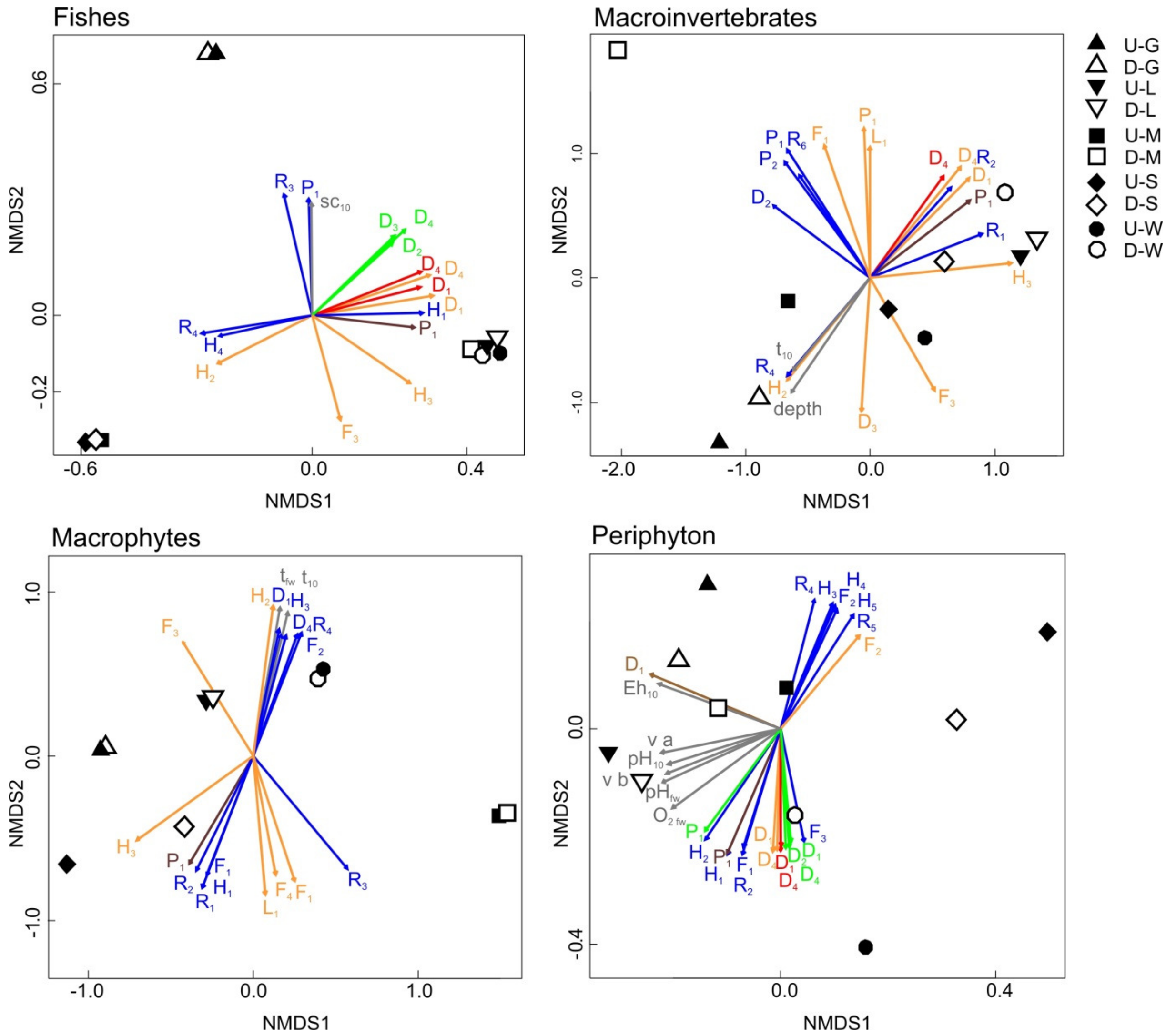


Fig. 7.5: Non-metric multidimensional scaling (NMDS) performed for different taxonomic groups, based on taxa abundance data and Bray Curtis similarity. Fishes: non-metric stress = 0.06 * 10^{-4}; Macroinvertebrates: non.metric stress = 0.03; Aquatic macrophytes: non-metric stress = 0.02; Periphyton: non-metric stress = 0.06. Study rivers are displayed with different pictograms, upstream sides (U) with filled symbols and downstream sides (D) with open symbols. Environmental variables and metrics (P ≤0.05 based on 1,000 permutations) are displayed as vectors and can be distinguished by colour according to their relatedness to physicochemical habitat characteristics (grey), fishes (blue), macroinvertebrates (orange), macrophytes (green), periphyton (brown) and all taxa (red) as well as by capital letters according to their relatedness to feeding type (F), locomotion type (L), reproductive strategy (R), productivity (P), diversity (D) and habitat requirements (H). Codes and coefficients of variance (r^2) of the environmental variables are shown in Table 7.4.

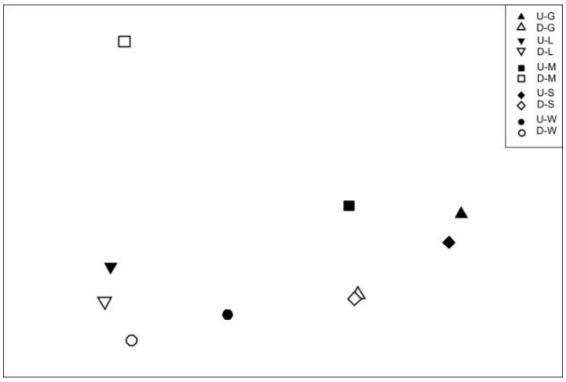


Fig. 7.6: Non-metric multidimensional scaling (NMDS) of the U and D sampling sides based on Euclidean distances resulting from biological traits and physicochemical habitat characteristics. Study rivers are displayed with different pictograms, upstream sides with filled symbols and downstream sides with open symbols, non-metric stress = 0.03.

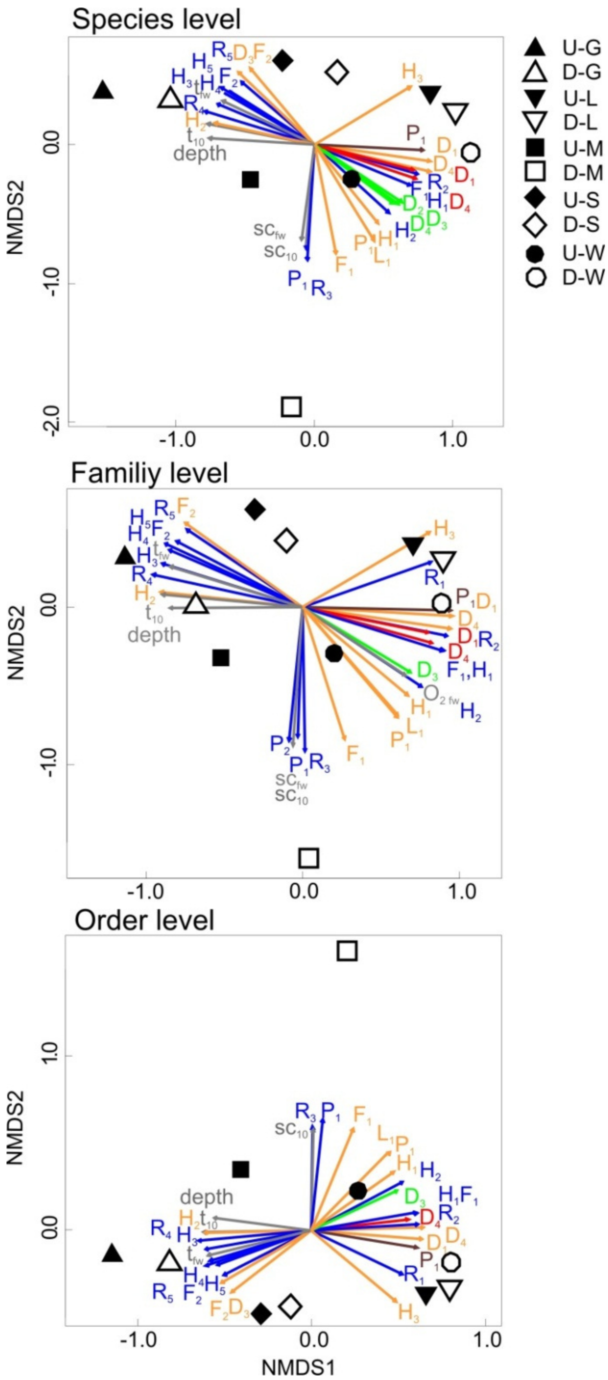


Fig. 7.7: Non-metric multidimensional scaling (NMDS) performed for different levels of taxonomic resolution, based on taxa abundance data and Bray Curtis similarity: species level = including all specimen that could be identified on species level, non-metric stress: 0.05; Families = including all specimen that could be determined to family level or further, summarized on family level, non-metric stress: 0.06; Orders = including all specimen summarized to order level, non-metric stress: 0.06. Environmental variables and metrics (*P* ≤0.05 based on 1,000 permutations) are displayed as vectors. For codes of study rivers, sides and environmental variables see legend Fig. 7.5 and Table 7.4.

The simultaneous inclusion of abiotic habitat variables and biological traits of all taxonomic groups in NMDS resulted in a more comprehensive and universally applicable assessment (Fig. 7.6). Both normalized distance-matrix based NMDS (including ecological traits and physicochemical variables of U and D as input variables) as well as classical NMDS and DCA (based on taxa abundance data, DCA not shown) revealed similar discrimination patterns of sides and rivers, indicating a strong linkage of ecological traits, community composition and habitat characteristics. For instance, the strongest separation of U and D was in both NMDS input scenarios found in the river M, and the weakest in the river L.

Generally, community effects were not only detectable on the species level, but also on higher levels of taxonomic resolution such as family and order level (Fig. 7.7, Tab. 7.4). Remarkably, differences upstream and downstream of weirs at adjacent sides

			R^2 taxonomic groups (Fig. 5)				R^2 taxonomic levels (Fig. 6)		
Group	Code	Full name	Periphyton	Macrophytes	Macroinvertebrates	Fishes	Species level	Familiy level	Order level
	D_1	Species richness	0.64 *			0.75 **	0.66 *	0.61 *	
All taxa	D_4	Hmax	0.68 *		0.63 *	0.84 **	0.73 **	0.68 *	0.64 *
	D_1	Species richness		0.65 *					
	D_2	Shannon Index			0.59 *				
	D_4	Hmax		0.61 *					
	F_1	Benthivor [%]	0.83 **	0.64 *			0.69 *	0.81 **	0.76 *
	F_2	Euryophag [%]	0.80 **	0.61 *			0.76 **	0.87 **	0.82 **
	F_3	Filtering [%]	0.60 ***						
	H_1	Rheophilic [%]	0.84 **	0.64 *			0.69 *	0.81 **	0.75 *
	H_2	High structural requirments [%]	0.82 **				0.64 *	0.77 *	0.72 *
Fishes	H_3	Indifferent [%]	0.84 **	0.66 *			0.70 *	0.82 **	0.77 *
	H_4	Fish Region Index	0.84 **			0.56 *	0.69 *	0.80 **	0.74 *
	H_5	No structural requirements [%]	0.80 **				0.65 *	0.76 **	0.67 *
	P_1	Total length [cm]			0.63 *	0.83 **	0.68 *	0.67 *	0.71 *
	P_2	Biomass [kg/100m3]			0.84 *			0.71 *	*
	R_1	Speleophilic [%]		0.78 *	0.58 *		0.60 .	0.70 **	0.68 **
	R_2	Rheopar [%]	0.79 **	0.64 *	0.59 *	0.74 *	0.74 *	0.82 **	0.77 ***
	R_3	Pelagophilic [%]		0.85 ***		0.94 ***	0.84 ***	0.78 ***	0.83 **
	R_4	Euryopar [%]	0.82 **	0.69 *	0.66 *	0.77 *	0.83 **	0.89 ***	0.85 *
	R_5	Phytolithophilic [%]	0.84 **				0.59 .	0.73 **	0.64 **
	R_6	Lithopelagophilic [%]			0.93 ***				
	D_1	Species richness	0.58 *						
	D_2	Shannon Index	0.64 *			0.71 *	0.56 *		
Macrophytes	D_3	Evenness				0.97 *	0.67 *	0.59 *	0.60 *
	D_4	Hmax	0.62 *			0.79 *	0.61 *		
	P_1	Abundance per side [%]	0.74 **						
	D_1	Species richness	0.69 *		0.79 **	0.91 ***	0.87 **	0.84 **	0.83 *
	D3	Evenness			0.72 **		0.63 *		0.67 **
	D_4	Hmax	0.67 *		0.82 **	0.93 ***	0.90 **	0.84 **	0.84 *
	F_1	Shredders [%]		0.68 **	0.79 *		0.78 *	0.72 *	0.80 *
	F_2	Undefined [%]	0.68 *				0.70 *	0.79 **	0.72 *
	F_3	Collectors [%]		0.71 **	0.67 *	0.72 *			
	F_4	Scrapers [%]			0.57 *				
Macroinver-tebrates	H_1	Rheophilic [%]					0.63 *	0.71 *	0.69 **
	H_2	Saprobic Index		0.90 **	0.69 **	0.70 *	0.75 **	0.77 **	0.77 **
	H_3	EPT%		0.81 **	0.80 **	0.86 **	0.80 **	0.81 **	0.81 **
	L_1	Active moving taxa [%]		0.75 *	0.69 *		0.77 **	0.78 **	0.82
	P_1	Individuals per sampling point			0.90 ***		0.81 **	0.78 **	0.79 **
	P_1	Cell count per cm2	0.70 *		0.64 *	0.64 *	0.76 **	0.83 **	0.76 **
Periphyton	D_1	Species richness [%]	0.91 ***	0.61 *					
	depth	Depth [m]			0.78 **		0.70 *	0.67 *	0.64 *
	O_{2fw}	Dissolved oxygen [mg/l]	0.82 **					0.56 *	
	sc_{fw}	Specific conductance free-flowing water [µS/cm]					0.57 *	0.72 **	
	sc_{10}	Specific conductance substratum [µS/cm]				0.77 *	0.68 *	0.63 *	0.69 *
Physicochemical Variables	t_{10}	Temperature substratum [°C]		0.87 **	0.58 *		0.66 *	0.75 *	0.75 **
	t_{fw}	Temperature free-flowing water [°C]		0.90 **			0.66 *	0.73 *	0.75 **
	Eh_{10}	Redox potential substratum [mV]	0.77 *						
	v a	Current speed above ground [m/s]	0.68 *						
	v b	Current speed below water surface [m/s]	0.68 *						
	pH_{fw}	pH of free-flowing water	0.77 **						
	pH_{10}	pH substratum	0.63 *						

Table 7.4: Codes of the environmental variables displayed in Figs 7.5 & 7.6 with full names and coefficients of variance (r^2) for environmental fitting in the NMDS for taxonomic groups (Fig. 7.5) and for taxonomic levels (Fig. 7.6).

within the same river were often greater than the differences observed among rivers from different geological units and drainage systems. For instance, differences on all levels of taxonomic resolution were greater between adjacent U and D sides at the river M than between the river M and the rivers G, S and W (Figs. 7.6 & 7.7) which belong to different drainage systems (G, M:Danube, S:Elbe, W: Main/Rhine) and which are geographically separated by more than 200 km (Fig. 7.1). On the other hand, the differentiation between the rivers L and W (Figs. 7.6 & 7.7) as well as between G and S was remarkably low in comparison to the other study rivers (Fig. 7.6), although these rivers are geographically separated by 300 km (W-L) and 400 km (G-S), belong to different drainage systems (G, L:Danube; W:Main/Rhine, S:Elbe) and to different geological units (G:molasses, L:limestone alps; W:chalkstone, S:basement).

The comparison of beta diversity (including all taxa) between U and D of individual rivers and between the rivers showed that the difference between U and D equals more than half of the differences between rivers (beta diversity between rivers: mean=0.68, beta diversity between sides: mean=0.35, Table 7.5). Variables which correlated with the ordination distances between study sides based on taxa distribution mostly refer to habitat preferences and functional feeding groups, as well as to diversity characteristics and physicochemical variables.

	βt				
River	Periphyton	Macroinvertebrates	Macrophytes	Fishes	All taxa
G	0.33	0.77	0.00	0.47	0.53
L	0.30	0.36	0.60	0.40	0.43
M	0.32	0.58	0.29	0.58	0.49
S	0.22	0.64	0.75	0.60	0.52
W	0.22	0.44	0.23	0.64	0.36

Table 7.5: Beta diversity (β_t) for each taxonomic group and for all taxa as measure of similarity between U and D side in each study river.

7.5 Discussion

The pronounced weir effects detected in this study suggest that damming strongly alters community structure, productivity and the diversity of stream ecosystems. These alterations are supposed to originate in an interruption of the natural gradient of physical habitat conditions and the biotic responses from the headwater to the mouth of river systems (Ward & Stanford 1983), as originally described in the River Continuum Concept (RCC) by Vannote et al. (1980). To our knowledge, this is the first study that comprehensively assesses the ecological effects of weirs on the serial river discontinuity including physicochemical habitat characteristics as well as community effects on all major taxonomic groups.

The most important finding is the overriding influence of weirs on biological communities compared to other variables including geology or drainage system. This finding was unexpected, since several authors suggest strong relatedness of rivers of the same or similar geological origin (Mykrä et al. 2007, Stendera & Johnson 2006, Kim *et al.* 2007) or drainage system (Corkum 1989, Richards et al. 1996, Robinson 1998, Schaefer & Kerfoot 2003). Consequently, the different geochemical conditions of the rivers included herein were expected to result in entirely different community structures. Remarkably, small scale effects of heterogeneity in *dg*, water depth and current speed introduced into adjacent sites of the same stream by weirs greatly exceeded the large scale effects of geology and geographic isolation.

Differences between taxonomic groups and rivers

This study shows that none of the single taxonomic groups (periphyton, macrophytes, macroinvertebrates, fishes) alone is a universally suitable indicator of the overall discrepancy in community structure upstream and downstream of weirs, yet they are widely used as indicators for the ecological status of aquatic ecosystems (e.g. macrophytes for the trophic status (Schneider & Melzer 2003), fishes for the ecological status in context of the EU-WFD (Dußling et al. 2005), periphyton for the ecological condition (Stevenson et al. 2008), macroinvertebrates for freshwater monitoring (Menezes et al. 2010)). The low congruence between the responses of different taxonomic groups to weirs is also supported by their individual and distinct responses to environmental gradients in other freshwater ecosystem studies (Heino 2010). For instance, Declerck et al. (2005) showed that different taxonomic groups in shallow lakes react individualistically to environmental gradients and Heino et al. (2005) revealed similar results for running waters. Based upon similarity values, diversity measures, functional traits and multivariate community composition analyses, none of the four taxonomic groups studied was a more integrative and sensitive indicator of weir effects than others. The signal strength of weir effects on biological communities turned out to be not only dependent on the taxonomic group investigated, but also differs strongly between rivers within taxonomic groups. This is mostly due to the stream-specific habitat structure, community composition, diversity and productivity which have strong influence on the discriminative power of different taxonomic groups (Heino 2010).

Periphyton

Periphyton, which constitutes the basis of aquatic food webs (Vannote et al. 1980, Szabo et al. 2008), strongly depends on physical habitat characteristics (Soininen 2002, Müllner & Schagerl 2003). This is supported by our data where most physicochemical variables revealed significant correlation with periphyton community composition. Along with the different abiotic habitat conditions observed in the study streams, this finding can explain the differing suitability of periphyton as an indicator of weir effects in different rivers. For instance, in the comparatively deep and slow flowing river Günz, cell counts of periphyton were 17-fold higher in the more shallow and high-current D compared to the U side. In the shallow and fast flowing Leitzach, periphyton cell counts were on average three-fold higher than in the Günz and only differed by a factor of 1.01 between D and U (Table 7.2).

Macrophytes

In contrast to periphyton, macrophytes only occurred in some streams and only one species discriminated between D and U which limits their use as a general indicator for weir effects. Only in two of the rivers (W, M) macrophyte diversity and abundance was high enough to detect differences between U and D, while differences in community composition were evident for the river S. However, as aquatic macrophytes can play an important role for habitat structure (Balanson et al. 2005), weir-induced alterations of the macrophyte community could affect the entire ecosystem in streams with high macrophyte dominance. This is evident from the strong correlation of macrophyte diversity measures with community composition of fishes and periphyton and of all taxonomic groups on species level.

Macroinvertebrates

Macroinvertebrate community structure (diversity indices, functional feeding groups, saprobial index, flow current preference, locomotion types) strongly discriminated between U and D sides which probably results from different flow conditions up-and downstream of weirs. Whereas the effect of the different flow velocities on flow current preference and locomotion types of macroinvertebrates is obvious (i.e. rheophilic and actively moving taxa being most characteristic for high-current D sides), current also affects the availability and ratio of coarse particular organic matter (CPOM) to fine particular organic matter (FPOM), which can explain the differences in functional feeding groups up- and downstream of the weirs. For instance, the higher abundance of the filter-feeding collectors Simulidae and Chironomidae at U sides with lower current may be explained by higher sedimentation rates of FPOM, and consequently higher FPOM/CPOM ratios. Accordingly, shredder organisms which are considered highly sensitive to perturbation (Rawer-Jost et al. 2000) were more abundant at D sides. Analogously, these effects on the distribution of functional feeding groups and organic matter seem to be also true for the general texture of the stream substratum which was much finer in U than in D. Fine-textures substrata typically reduce the availability of voids and consequently the abundance and diversity of benthic organisms in the hyporheic zone (Gayraud & Phillipe 2003, Geist & Auerswald 2007, Rice et al. 2010), which can explain the lower abundance and diversity of Plecoptera, Ephemeroptera and Trichoptera at U sides.

Fishes

The observed differences of up to two fish faunal regions (according to the classification by Dußling et al. 2005) between U and D mirror community structures with entirely different ecological requirements and represent habitat conditions typical for rhithron vs. potamon regions within entire stream ecosystems. Weir-associated fish habitat modifications mostly result from changes of water depths, current speed and substratum composition, which compared to the natural status are more pronounced in U than in D. In most cases, U sides cannot fully meet the habitat requirements of species with high structural requirements but are tolerated by indifferent species (Kruk 2007). For instance, the rheophilic species *S. trutta*, *C. gobio*, *G. gobio* and *S. cephalus* occurred at higher densities in D sides, whereas the generalist species *R. rutilus* occurred in higher densities in U sides. In rivers with occurrence of high numbers of specialised (e.g. rheophilic, lithophilic or benthivoric) fish species (e.g. M), the most pronounced differences in fish community structure between U and D were observed, while differences in rivers with a high number of tolerant species where generally low (e.g. G).

Implications for management

The continuity of river systems is a hydromorphological quality element in the European Water Framework Directive (WFD) which requires the evaluation of human impacts on water bodies (European Parliament 2000). However, river sections in spatial proximity of weirs are currently excluded from assessments in context of the WFD. As most European rivers today are a mosaic of upstream and downstream sides of weirs that succeed each other in short geographical distances, information on the qualitative and quantitative effects of weirs on these river sections is crucial for representative assessment of their ecological status and for conservation and restoration management. For example, restoration measures which form a variety of shallow overflowed habitats could improve the overall biodiversity in weir-regulated rivers with increased and uniform water depths and reduced current speed (Freeman et al. 2001, Kemp et al. 1999). The normalized NMDS based on physicochemical variables and ecological traits is highly suitable for a comprehensive quantification of weir effects in different rivers on the ecosystem level as well as for the monitoring of restoration measures. Additionally, this method provides the possibility to assign indicator weights to specific target taxa or ecological traits to account for conservation management prioritization.

Due to river-specific differences, the univariate consideration of single abiotic parameters and of single taxonomic groups is not suitable as a generally applicable indicator for the detection of weir effects. Multivariate methods which simultaneously include different taxonomic groups and physicochemical variables produce a more complete and coherent picture of the serial discontinuity and may serve as a comprehensive and universal indicator of ecosystem health.

Community effects and the underlying effectors were generally detectable at high levels of taxonomic resolution such as family and order level. This illustrates that the effects of the interruption of the river continuum caused by weirs are not only restricted to a few sensitive species or taxonomic groups but affect the entire aquatic community structure. Therefore, a classification on the family or even order level may be sufficient for most taxonomic groups. This finding is particularly relevant for the applicability of this methodology in other regions with different community composition. Typically, funding for the ecological monitoring of weir effects and of other impacts on aquatic ecosystems is limited. The results of this study suggest that the inclusion of multiple taxonomic groups at low levels of resolution is advantageous compared to the inclusion of few groups at high levels of taxonomic resolution in ecological monitoring.

8 Synthesis: a new integrative approach to improve the course of action for river restoration and the efficiency control of restoration measures

Methods of active and passive bioindication as applied in the presented case studies revealed a high functional reliability and effectiveness to detect anthropogenic and restoration induced changes in stream ecosystems. Target species- and life stage-specific tests applying active bioindication can be combined with single group or multi group passive bioindication analyses. For a clear scientific picture of the pre-restoration and post-restoration conditions, these biotic components should be coupled with assessments of abiotic physicochemical habitat variables and analysed through univariate and multivariate statistics. Using the integrative power of bioindication assessments enables the detection of restoration-induced changes even if funding is limited or the assessment of the pre-restoration status is not possible (e.g. in newly built flow courses or if the monitoring starts after the restoration measures were carried out). For a better understanding of the complex interactions between ecosystem processes and restoration measures which determine river restoration success, the monitoring of projects has to be extended, at least to a subset of pilot studies being assessed.

8.1 Challenges of using bioindication for measuring the success of stream restoration

Aquatic indicator organisms for bioindication

For active bioindication purposes with algae and macroinvertebrates, the availability of sensitive indicator organisms is so far restricted to their use for ecotoxicological questions. These organisms are mainly used for the detection of lethal effects of environmental pollutants or water quality (Fleeger et al. 2003). Water quality can also be an important question for the restoration of river ecological function, but today structural degradation is also a crucial factor in European streams. However, most ecotoxicological indicators are of limited use for the detection of structural habitat improvements which can determine the success of habitat restoration. Structure-sensitive macroinvertebrate and algae species are currently not produced under standardized conditions for habitat assessments in sufficient numbers. Due to their distinctive habitat requirements, fish in different life stages are sensitive to water quality, substratum quality and structural degradation (Jungwirth et al. 2003, Kottelat & Freyhof 2007). In each fish region there are fish species with highly adapted life cycle strategies resulting in a narrow ecological amplitude and making them very sensitive to environmental changes. In particular, the extreme habitat conditions in headwater streams with strongly fluctuating discharges, high current speed and substratum dynamic have evolved species with complex life cycle strategies. These species have often highly specialised and distinctive adaptions to feed, to reproduce, to manage discharge extremes and to cope with climate induced seasonal changing habitat conditions. Especially all salmonid fishes and in particular their early life stages are water and substratum quality-sensitive (Rubin & Glimsäter 1969, Soulsby et al. 2001b, Crisp 1996). The adult stages need clean, cool and fast-flowing water with high oxygen content (Kottelat & Freyhof 2007). Egg and larval stages are strongly dependent on coarse river bed substratum comprising high hyporheic exchange rates which ensure a high oxygen supplement and the exchange of metabolites (Jungwirth et al. 2000, Jungwirth et al. 2003). Juvenile stages need shallow bank habitats with low current speed and a finer substratum composition (Jungwirth et al. 2003). Due to the complex habitat requirements, population analysis of *Salmo trutta* as a passive indicator can be used to assess the ecological integrity of headwater sections. Single life stages, such as eggs or larvae, can be used to test the quality of specific habitats like river bed substratum or restoration effects of spawning grounds using active bioindication. Due to their high economic value for consumption or sport fishing (e.g. *Salmo trutta* and *Salmo salar*) these fish species are intensively produced in aquaculture. Consequently, eggs, juveniles and adults are easily available in high numbers and standardized quality which is also an important criterion for the suitability as active bioindicators. In contrast, the production of other potential bioindicators, such as freshwater mussels, is still a challenge.

Indicator systems for bioindication (active bioindication)

For the use of salmonid eggs in field surveys to test water or substratum quality, indicator systems of high reliability for the exposure are necessary. The material and construction scheme should allow a sensitive detection of changes in environmental gradients without impacting the development of the indicator organisms. To detect which factors contributed most to the assessment status or influenced the exposed organisms it is essential to gather detailed information about physicochemical changes during the exposure time. For this reason it is necessary to equip indicator exposure systems with measurement units to allow a variety of spatial resolved measurements. This can be particular interesting in investigations of spawning grounds where the eggs can naturally be buried by the salmonids in different depth layers and metabolic rates can have a strong influence on the hatching success (Soulsby et al. 2001b). The linkage of survival rates and stages from the exposed indicators in combination with waterchemical variabels and physical measurements like substratum texture can produce a detailed picture of ecological processes which all account to the detection of the effectiveness of restoration measures. The “salmonid-egg floating box” (SEFLOB, Pander & Geist 2010b) and the “egg sandwich” (ES, Pander et al. 2009) were developed for this purposes. Both toolboxes showed a high reliable functionality and cost effectiveness. The SEFLOB was used in a pre-restoration assessment for the re-introduction of the highly endangered Danube salmon in rivers which formerly served as spawning habitats. In this study the SEFLOB approved to be much more sensitive for the detection of deficiencies in water quality that are limiting for egg survival than chemical measurements. This demonstrates the suitability of the SEFLOB to test if water chemical conditions for the target species are sufficient before an implementation of restoration measures or the reintroduction of the target species is carried out. The results of the “egg sandwich” in artificial and natural spawning grounds of *Salmo trutta*, and *Thymallus thymallus* (Pander et al. 2009) suggest that this indicator system is an easy and cost effective tool to evaluate the ecological functionality of the streambed and the effectiveness of spawning ground restoration. The system is applicable for the evaluation of all types of substratum restoration which are currently practiced, such as gravel introduction, raking, gravel washing and to test sites of structural improvements to induce gravel relocation. The system also has the potential to be adapted for assessments in various biogeographic or fish ecological regions by testing with different indicator organisms as other fish species (e.g. rainbow trout (*Oncorhyncus mykiss*) in North America) or macroinvertebrates (e.g. waterquality and substratum quality sensitive organisms such as thick shelled river mussel (*Unio crassus*)).

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 to date. Well-established indicators and exposure systems for the evaluation of the effects of measures such as the introduction of dead wood, macrophytes, boulders, shallow habitats, riparian wood or fish bypass channels are widely lacking. Due to the ambiguous spatial effectiveness and the structural complexity of these measures it is probably very 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) with stocked or naturally occurring individuals which can freely move between their preferred habitats can deliver insights in the functionality of stream restoration measures (Pander et al. 2011). Organisms with high structural requirements like freshwater crayfish (*Astacus astacus*, Lundberg (2004)) or bullhead (*Cottus gobio* Kottelat & Freyhof (2007)) have great potential for the detection of structural improvements with this method. Additionally, the differences in habitat preference of stocked versus wild fish (e.g. *Salmo trutta* in Pander et al. (2011)) can be used for a broad field of specific research questions on autecology dependent restoration success.

The suitability of passive bioindication

Ecosystem assessments typically contain the investigation of numerous variables. Since only view variables can be responsible for the detection of processes or improvements (which at the beginning of the project are often unknown or not detectable) and a comprehensive ecosystem monitoring mostly fails on limited funding it can be very effective to investigate factors which are highly integrative. Therefore bioindication of one or more levels of biological organisation (depending on the research question adressed) can be very advantageous (Mueller et al. 2011).

With increasing complexity of the restoration effort (Figs. 1.1 and 8.2), passive bioindication can be advantageous because several combined measures and their ecological interactions can be evaluated comprehensively. A limitation using traditional bioassessment methods (Table 1.3) in strongly altered or restored habitats can be the restricted species inventory (Noss et al. 1995, Pander & Geist 2010a, Pander et al. 2011), high numbers of ubiquitous species (Kirchhofer 1995, Pander & Geist 2010a) and increasing numbers of neobiota (Pyšek & 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 for assessments of the pre- and post-restoration status if abundances of indicator taxa are very low. In this case the reduction of data to single indices as practiced for the WFD cannot be recommended for the evaluation of restoration success. Additionally, in restoration programs of transnational scale or for a 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 passability, removal of weirs and bank reinforcement or the construction of nature like river courses have to be implemented in different geographical regions (e.g. Danube wide) or scales (tributary to catchment).

The suitability of multivariate analysis of passive bioindication

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). The relative comparison of habitat characteristics, community composition and functional traits between restored and unrestored sites allows the quantification of the effects of different bank habitat restoration measures (Pander & Geist 2010a) or the assessments of the functionality of fish bypass channels as compensatory habitats (Pander et al. 2011). The methodology assessing the present fish community in several replicates of standardized length or area per treatment (restoration measure, habitat type) turned out to be suitable to distinguish habitats, detect seasonal effects on colonization of habitats and to draw conclusions about restoration success and remaining deficits on ecosystem level. Furthermore this data analysis strategy integrates biotic and abiotic data and is basically independent of the sampling method, the variables which were assessed, the river specific species composition and the occurrence of indicator taxa or the target species of restoration.

Passive bioindication is also applicable to detect major anthropogenic impacts e.g. the disruption of the river continuum (Mueller et al. 2011) which can be essential to point out main impact factors (weirs) and most limiting deficiencies (degradation of habitats and change in biodiversity) in the pre-restoration proceeding. Since the response of biological communities to changing environmental conditions can be inconsistent between taxonomic groups and rivers (Johnson et al. 2006, Paavola et al. 2006, Heino 2010, Mueller et al. 2011), assessments on ecosystem scale should include several trophic levels of animals and plants. The combination of several levels of biological organization can reveal even small scale effects as weir introduced heterogeneity in water depth, substratum composition and current speed (Mueller et al. 2011). The challenge thereby is to combine different data structures which result from different sampling strategies and scales. For instance, freshwater algae can be scraped off from a small stone while fishes have to be sampled with fish traps, net-fishing or electrofishing from a rather large spatial area. Additionally data often differ in taxonomic resolution and the degree of accuracy (presence absence data, relative abundances and quantitative data). The multivariate approaches like the multi-dimensional scaling (none metric NMDS, metric MDS) which were used in Pander & Geist (2010a), Mueller at al. (2011) and Pander et al. (2011) can be applied for different data structures and for each taxonomic group.

8.2 New ways for an integrative assessment of target species oriented restoration success and overall river ecological functioning

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 the restoration effects. A stepwise evaluation of the main impact factors of disturbance or degradation with increasing complexity from water quality assessments to the evaluation of river ecological function may be most suitable to consider all major drivers of a successful restoration by simultaneously focusing stepwise on accuracy and extent of required information. As river restoration can affect ecosystems in many ways, there is seldom one, ideal scale at which to conduct an ecosystem assessment that will suit several purposes (Millenium Ecosystem Assessment 2005). Restoration induced changes can have small scale effects like the improvement of life stage-specific habitats of single species or large scale effects like population dynamics and interactions between different levels of biological organization or the restoration of ecosystem services. To cope with the challenges of determining scale and intensity of the investigations (water quality, structural

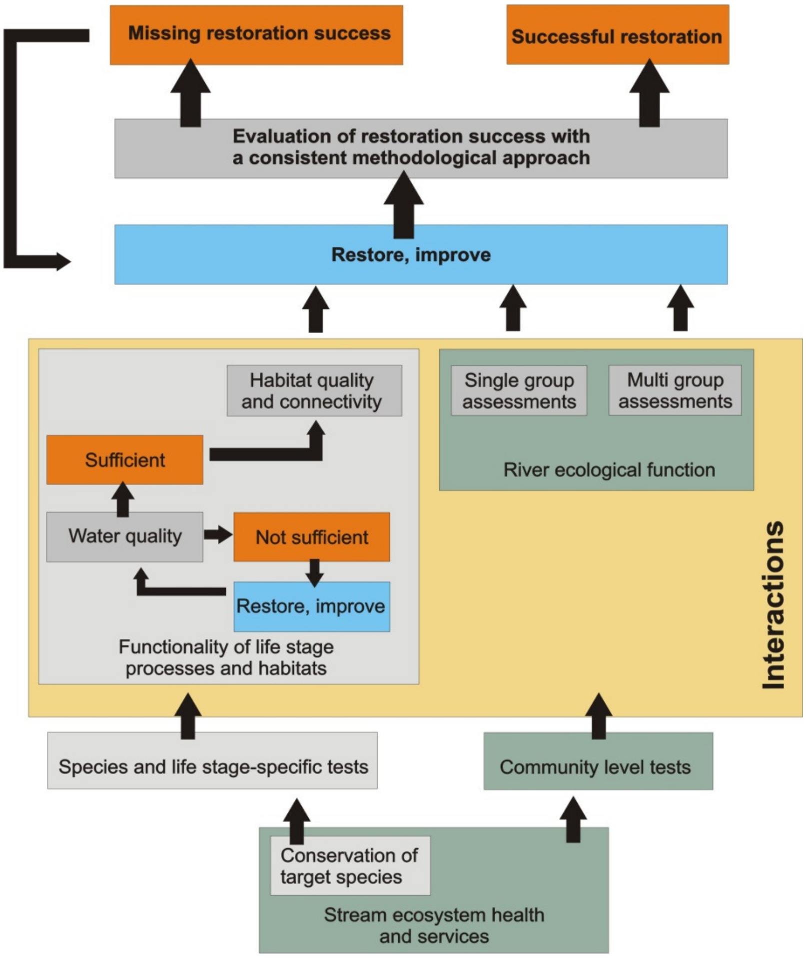


Fig. 8.2: Flow chart of an integrative efficiency control of restoration measures or ecosystem assessments. Black arrows indicate a step by step approach. The assessment strategy is subdivided in a target species focused part (highlighted in grey) and a second part for the assessment of ecosystem health and ecosystem services (highlighted in green). The combination of assessment methods of different complexity allows the investigation of interactions between different levels of organization (highlighted in yellow). The results of the assessment steps (highlighted in orange) can lead to different restoration measures and actions (highlighted in blue) or a return to previous steps.

habitat quality, single group versus multi group, taxonomic resolution) 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. 8.2). For instance, the strong decline of the endangered salmonid *Hucho hucho* (Geist et al. 2009) 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 of the food web. All these factors can contribute to a reduced reproductive fitness and a drop down of the population below a critical size (Allee- effect, minimum viable population (Allee 1931)). To point out the most limiting factors for salmonid conservation and the restration process, a methodological approach should at first test the general sufficiency of the water quality of the study stream (Pander & 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 & Geist 2010b) should be applied again to test for potential improvements (Figs. 8.2

and 8.3). If water quality requirements are fulfilled life stage-specific strategies and key-habitats should be under focus next. This includes the assessment of spawning ground suitability (Pander et al. 2009), juvenile habitats, adult habitats and the habitat connectivity (Pander & Geist 2010a, Denic & Geist 2010, Pander et al. 2011,) which is important for spawning migrations and movements between life stage-specific habitats (Geist et al. 2009). As *Hucho hucho* is a top predator in stream ecosystems (Holčík 1990) with naturally low species abundance the quality of species specific habitats may be not 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 in 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 tests for the target species *Hucho hucho* (Geist et al. 2009), as suggested in the IFEBC-concept (Geist 2011), can determine the fitness of the remaining individuals on population level by analysing the degree of inbreeding what can be necessary if breeding programs and stocking for species reintroduction is an option. To choose the best population for reintroduction the SEFLOB method and the “egg-sandwich” can also be applied. With these systems it is possible to find out which population can cope best with the distinctive environmental conditions of the study stream in their most critical life stages. To improve the applicability of these tools for bioindication the coordination of the river restoration procedure should be standardized and the exchange of information between scientists, restoration experts, decision makers and stakeholders should be much more intensive.

8.3 A standardised restoration protocol “The Proceeding Chain of Restoration”

The presented new tools for bioindication can be most effective when they are integrated in a standardised protocol for river restoration which leads to a target-oriented improvement of species conservation as well as overall ecosystem health.

Improved knowledge about the quality and functionality of restoration measures and a standardized implementation and monitoring could be achieved by following a target-oriented restoration protocol which is proclaimed here as the “Proceeding Chain of Restoration” (PCoR, Fig. 8.3). The PCoR is a step by step approach which systematically structures the complex procedure of river restoration. It is subdivided into three general parts, the pre-restoration proceedings, the implementation of measures and the post-restoration proceedings.

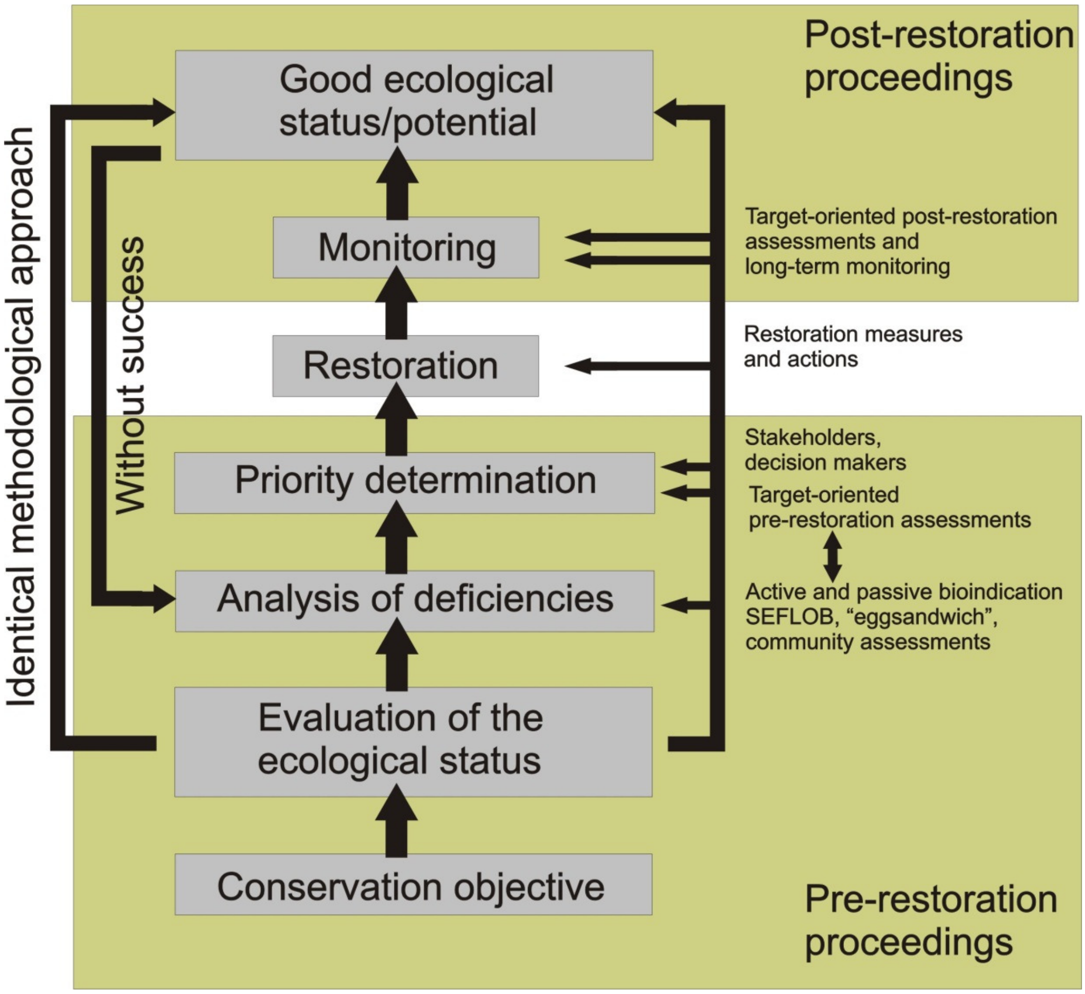


Fig. 8.3: The “Proceeding Chain of Restoration” (PCoR) is a step by step approach which systematically structures the complex procedure of restoration from pre-restoration proceedings, restoration measures and actions to post restoration proceedings. Pre- and post-restoration proceedings are highlighted in green. The single steps of the proceeding chain are highlighted in grey.

The pre-restoration proceeding

As first step of the pre-restoration proceeding, a clear **conservation objective** has to be defined. A pinpointed conservation objective is indispensable to know the items and the scale at which these have to be assessed in the next step and will be very useful to set priorities (Dahm et al. 1995). A clear conservation objective also gets important when it is necessary to focus on one target species out of a pool of several species of high conservation value which compete for resources and space (Simberloff 1998).

The **evaluation of the pre-restoration status** is a key stage in river restoration. It is the basis for the analysis of deficiencies and the priority determination were the most effective measures will be discussed by the involved stakeholders. Ideally the investigation of the pre-restoration status covers a broad variety of aspects contributing to overall river ecological function including the serial discontinuity (Mueller et al. 2011) as well as habitat alterations (Pander et al. 2011) and water quality assessments (Pander & Geist 2010b).

The restoration of river ecological function can address a broad variety of aspects on different scales e.g. riparian management, floodplain reconnection, water quality aspects, bank stabilization, dam removal, fish passability and channel reconfiguration (Kondolf et al. 2007). Consequently the evaluation of river restoration success has to be adjustable to the purpose of the restoration measure. For instance, the assessment of overall ecosystem health (typically on larger spatial scale) requires a different approach than the assessment of life stage-specific restoration like the improvement of spawning ground quality for salmonids or the mitigation of migration barriers (typically on spatial small scales).

Since habitat conditions within small rivers can be very patchy and highly variable (Poff & Ward 1990, Winemiller et al. 2010), it can be necessary to evaluate the effects of river restoration in a comprehensive approach. Increased variability can complicate the determination of restoration effects and requires additional knowledge of the ecosystem status particularly before and after the restoration (Chapman 1999). Concerning initial stages 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 the following changes.

Additionally different levels of biological organization can have divergent recovery rates and time (Power 1999) and so the detection of failure or success can be linked to the developmental stage of the measure at the monitoring timepoint. Therefore it is highly recommended to assess several levels of biological organization when only a short-term monitoring is carried out or to consider at least the successive status of the restoration site in the evaluation scheme. On ecosystem scale there are naturally complex interactions within species, between species and between species and habitats which are all based on the different sensitivity of species or life stages to associated ecosystem processes (Clarke & Warwick 2001). Processes on ecosystem level naturally need time to develop important ecosystem functions for a stabilized and self-sustainable system. Short-term monitoring actions can detect all direct effects of restoration actions, but complex interactions can only be detected in long-term assessments.

In the project stage of evaluating the deficiencies, standardised methods and applications (Bernhardt et al. 2005) will be specified as a framework for a continuously applied efficiency control of the implemented measures and actions in the following proceedings.

Analysing the deficiencies and evaluating the most determining factors responsible for the degradation leads consequently to the proposal of effective measures and actions. As for the methods and applications to evaluate the pre-restoration status, the applied statistics for the data analyses can be determined in a framework for the further proceeding. For example, the sampling design (e.g. number of samples, length of river stretch), the degree of data transformation (e.g. square-root transformation, log-transformation) and the statistical methods (e.g. non-metrtic multidimentional scaling, principle component analysis) can be specified in a standardized protocol for application in the post-restoration proceedings.

In the stage of the **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 several options concerning contrary restoration measures or conflicts between target species.

The post-restoration proceeding

Every post-restoration status keeps the potential to evaluate which measure contributed most to the compliance of the conservation objective at the beginning (Downs & Kondolf 2002). Thereby it is not necessarily important to evaluate all projects for an enhanced knowledge of the functionality of restoration measures. In some cases a subset of so called “pilot studies” may be sufficient if the same measurement types in comparable rivers are considered (Bernhardt et al. 2007). Ideally the subsets cover the assessment of a broad range of restoration types, e.g. restoration of water quality (Pander & Geist 2010b) bank habitat restoration (Pander & Geist 2010a), spawning ground restoration (Pander et al. 2009) and fish passability (Pander et al. 2011). To determine the restoration success measurement strategies, assessment tools and applications for the data analyses, which were carried out in the pre-restoration stage should be applied again following a consistent methodology.

To consider the post-restoration status with a consistent methodological approach can avoid data failures and misinterpretations and enables to distinguish restoration effects from other ecosystem processes. This is a fundamentally important fact to measure the strength and direction of how implemented measures are influencing habitats or ecosystem. The knowledge of the main drivers of successful restoration will avoid trial and error based proceedings and can lead to systematic target-oriented stream restoration.

According to the proceeding chain of restoration, all restoration efforts should be followed by a post-restoration monitoring, where the effectiveness of the measures is evaluated and published (regionally, nationally and internationally, Nienhuis & Gulati 2002). This enables restoration experts to profit from the findings and to improve further restoration planning. To comply with the European directive “Public access to environmental information” (Table 1.1) and for the involvement of a wider public audience, expert knowledge should be collected, transformed into a generally understandable form and presented on easy accessible platforms (e.g. web-sites of national conservation authorities).

In the monitoring of the post-restoration proceeding, short-term and long-term investigations can be applied to evaluate the success of restoration measures. The temporal and spatial resolution and the number of investigated organism groups in the post-restoration surveys should be adapted to the restoration goal (e.g. structural improvements or improvement of overall ecosystem health) and the methodology should be kept constant from the pre-restoration survey on. This was also proclaimed by Bernhardt et al. (2007), who observed that the extent of the monitoring program is more influenced by limited project funding than it is adapted to a specific monitoring question. Ideally, there should be a strong linkage between project managers, restoration experts and scientists to fulfill the criteria of the PCoR in future restoration projects. In many cases restoration measures are implemented without an assessment of the pre- or post-restoration status (Table 1.2), causing limited restoration success and learning effect. 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 in this study, in combination with assessments of ecosystem processes can fulfill the criteria of the CARE principle.

8.4 Recommendations for future research

In Europe almost all stream ecosystem types are affected by human alterations. The caused ecosystem degradations reach from the headwaters to the estuaries and include areas of high as well as low biodiversity similarly. Each river ecological region has evolved highly adapted species guilds comprising different levels of biodiversity. Since high biodiversity is no quality component for ecological integrity (concerning fishes, all headwaters typically have species communities of low biodiversity), river regions with high as well as river regions with low biodiversity are of conservation interest. Consequently, all types of stream ecosystems should be considered in future research.

According to the high variability of river types, there is a large number of measures to restore anthropogenic induced degradations. To improve the standardized implementation of the PCoR-principle it is necessary to enhance future research efforts to test the functionality of the restoration measures. Since restoration success is basically dependent on the knowledge about ecological requirements of target species and the knowledge about ecosystem health determining processess it will be important to intensify research on the autecology of species as well as research about the ecological functionality of stream ecosystems. Much more information is required about the autecology of fish, macroinvertebrates, macrophytes, algae and microorganisms to understand and restore highly specialized processes of life cycles from microhabitat to macrohabitat scale. This is particularly important for all flagship, umbrella or target species which play a key-role as indicators for overall ecosystem health. For instance, the juveniles of the highly endangered freshwater pearl mussel *Margaritifera margaritifera* spend several years in the river bed substratum with distinct physicochemical habitat requirements (Geist 2010). However, little is known about the essential components of their microbial diet which are necessary for a successful juvenile developement. Microbial processes in the interstitial zone determine the functionality of the stream bed substratum (Hancock et al. 2005). These processes are strongly linked to the integrity of the food web and overall stream ecosystem health (Fischer et al. 1996, Findlay 2010). Also the endemic Danube salmon which is an apex predator, ecosystem engeneer, indicator- and flagship species in the Danube system (Geist et al. 2009) is dependent on the integrity of the food web. However, details about required fish biomass concentration to allow the successful switch from yolk sack diet to natural prey in habitats of his most critical life stage are still unknown. A better understanding of these functional relations between organisms and their habitats can leed to a more effective restoration of life stage specific habitats which also contribute to overall ecosystem health (improvement of the integrity of the food web). In this context future research should include questions such as the homeranges of species, minimum required populations, minimum required habitat space (also habitats for single life stages), structural habitat provisions and the required productivity as well as the integrity of the food chain. Furthermore, knowledge about the autecology of species is the basis to establish new indicator organisms for active and passive bioindication assessments.

Traditional restoration measures like channel reconfiguration, the implementation of structural improvements (e.g. boulders, dead wood and groynes) and the restoration of the river bed substratum should be tested for their contribution to the improvement of autecology dependent habitat requirements of species. This can be done in a similar way like it was done for spawning ground restoration, the assessment of bank habitats or the investigation on weirs and fish bypass channels in this study.

To quantify the effects of further restoration measures, the methods presented in this study can be used, but future research is needed to adjust them to the particular requirements. For the development of comprehensive and at the same time cost effective monitoring strategies, enhanced knowledge about the reactions of multiple taxonomic groups on restoration induced changes, the minimum required taxonomic resolution and the best spatial and temporal scale for the investigations is needed. Since many restoration projects suffer on limited funding it is always important for managers and stakeholders to keep projects as cost efficient as possible. Choosing the adequate scale for the assessment and the best required taxonomic resolution for species determination has a great potential to spare costs without losing to much information quality.

In general, a combination of basic research on autecology questions and applied investigations about the drivers, which determine the ecological functionality of restoration measures, promise the most efficient results to expand current knowledge to other target species, geographic regions and restoration types.

9 References

Aarts B.G.W., Van Den Brink F.W.B. & Nienhuis, P.H. (2003). Habitat loss as the main cause of the slow recovery of fish faunas of regulated large rivers in Europe: the transversal floodplain gradient. *Regulated Rivers: Research & Management* **20**, 3-23.

Acornley R.M. & Sear D.A. (1999). Sediment transport and siltation of brown trout (Salmo trutta L.) spawning gravels in chalk streams. *Hydrological Processes* **13**, 447-458.

Agostinho A.A., Gomes L.C., Fernandez D. R.. & Suzuki H.I. (2002). Efficiency of fish ladders for neotropical ichthyofauna. *River Research and Applications* **18**, 299-306.

Allee W.C. (1931). *Animal Aggregations. A Study in General Sociology.*University of Chicago Press, USA.

Almeida E.F., Oliveira R.B., Mungai R., Nessimian J.L. & Baptista D.F. (2009). Effects of small dams on the benthic community of streams in an Atlantic forest area of Southeastern Brazil. *International Review of Hydrobiology* **2**, 179-193.

Angermeier P.L. & Karr J.R. (1984). Relationships between woody debris and fish habitat in a small warmwater stream. *Transactions of the American Fisheries Society* **113**, 716-726.

Armitage P.D., Moss D., Wright J.F. & Furse M.T. (1983). The performance of a new biological water quality source system based on macroinvertebrates over a wide range of unpolluted running-water sites. *Water Research* **17**, 333-347.

Arrhenius O. (1921). Species and Area. *Journal of Ecology* **9**, 95-99.

Baker C.F. & Boubée J.A.T. (2006). Upstream passage of inanga *Galaxias maculatus* and redfin bullies *Gobiomorphus huttoni* over artificial ramps. *Journal of Fish Biology* **69**, 668-681.

Balanson S., Walton B.M., Wolin J.A. & Mal T.K. (2005). Aquatic Macrophyte Diversity and Habitat Characterization of the Cuyahoga River Watershed in Northeastern Ohio. *The Ohio Journal of Science* **105**, 88-96.

Balian E.V., Lévêque C., Segers H. & Martens K. (2008). The freshwater animal diversity assessment: an overview of the results. *Hydrobiologia* **595**, 627-637.

Barbour M.T., Gerritsen J., Snyder B.D. & Stribling J.B. (1999). *Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, second edition.* EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C., USA.

Baumgartner L.J. & Harris J.H. (2007). Passage of non-salmonid fish through a deelder lock on a lowland river. *River Research and Applications* **23**, 1058-1069.

Beierkuhnlein C. (2003). Der Begriff Biodiversität. *Nova Acta Leopoldina* **328**, 51-71.

Bellinger E.G. & Sigee D.C. (2010). *Freshwater Algae: Identification and Use as Bioindicators*. John Wiley & Sons, West Sussex, United Kingdom.

Bernhardt E.S., Palmer M.A., Allan J.D., Alexander G., Barnas K., Brooks S., Carr J., Clayton S., Dahm C., Follstad-Shah J., Galat D., Gloss S., Goodwin P., Hart D., Hasset B., Jenkinson R., Katz S., Kondolf G.M., Lake P.S., Lave R., Meyer J.L., O'Donnell T.K., Pagano L., Powell B. & Sudduth E. (2005). Synthesizing U.S. River Restoration Efforts. *Science* **308**, 636-637.

Bernhardt E.S., Sudduth E.B., Palmer M.A., Allan J.D., Meyer J.L., Alexander G., Follstad-Shah J., Hasset B., Jenkinson R., Lave R., Rumps J. & Pagano L. (2007). Restoring rivers one reach at a time: results from a survey of U.S. river restoration practitioners. *Restoration Ecology* **15**, 482-493.

Bernier-Bourgault I., Guillemette F., Vallee C. & Magnan P. (2005). A new incubator for assessment of hatching and emergence success as well as the timing of emergence in salmonids. *North American Journal of Fisheries Management* **25**, 16-21.

Bizzotto P.M., Godinho A.L., Vono V., Kynard B. & Godinho H.P. (2009). Influence of seasonal, diel, lunar, and other environmental factors on upstream fish passage in the Igarapava Fish Ladder, Brazil. *Ecology of Freshwater Fish* **18**, 461-472.

Bohl M. (1999). *Zucht und Produktion von Süßwasserfischen*. 2nd edn. München: Verlags Union Agrar VUA, Germany.

Bratrich C., Truffer B., Jorde K., Markard J., Meier W., Peter A., Schneider M. & Wehrli B. (2004). Green hydropower: a new assessment procedure for river management. *River Research and Applications* **20**, 865-882.

Bredenhand E. & Samways M.J. (2009). Impact of a dam on benthic macroinvertebrates in a small river in a biodiversity hotspot: Cape Floristic Region, South Africa. *Journal of Insect Conservation* **13**, 297-307.

Brunke M., Hoffman A. & Pusch M. (2001). Use of mesohabitat-specific relationsships between flow velocity and river discharge to assess invertebrate minimum flow requirements. *Regulated Rivers: Research & Management* **17**, 667-676.

Buddensiek V., Engel H., Fleischauer-Rössing S., Olbrich S. & Wächtler K. (1990). Studies on the chemistry of interstitial water taken from defined horizons in the fine sediments of bivalve habitats in several northern German lowland waters. *Archiv für Hydrobiologie* **119**, 55-64.

Bundesministerium der Justiz (2009). Gesetz zur Ordnung des Wasserhaushalts (Wasserhaushaltsgesetz - WHG). *BGBl.* **I**, p. 2585.

Bunn S.E. & Arthington A.H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* **30**, 492-507.

Calles O. & Greenberg L. (2009). Connectivity is a two-way street – the need for a holistic approach to fish passage problems in regulated rivers. *River Research and Applications* **25**, 1268-1286.

Camusso M., Balestrini R., Muriano F. & Mariani M. (1994). Use of freshwater mussel *Dreissena polymorpha* to assess trace metal pollution in the lower river Po (Italy). *Chemosphere* **29**, 729-745.

Chapman M.G. (1999). Improving sampling designs for measuring restoration in aquatic habitats. *Journal of Aquatic Ecosystem Stress and Recovery* **6**, 235-251.

Clarke K.R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**, 117-143.

Clarke K.R. & Warwick R.M. (2001). *Change in marine communities: an approach to statistical analysis and interpretation*, 2nd edition.PRIMER-E, Plymouth, United Kingdom.

Copp G.H. (1992). An empirical model for predicting microhabitat of 0+ juvenile fishes in a lowland river catchment. *Oecologia* **91**, 338-345.

Corkum L. (1989). Patterns of benthic invertebrate assemblages in rivers of northwestern North America. *Freshwater Biology* **21**, 191-205.

Cox E. (1996). *Identification of Freshwater Diatoms from Live Material.* 1st. Edition. Chapman & Hall. London, United Kingdom.

Crisp D.T. (1996). Environmental requirements of common riverine European salmonid fish species in fresh water with particular reference to physical and chemical aspects. *Hydrobiologia* **323**, 201-221.

Dahm C.N., Cummins K.W., Valett H.M. & Coleman R.L. (1995). An ecosystem view of the restoration of the Kissimmee River. *Restoration Ecology* **3**, 225-238.

Daugherty D.J. & Sutton T.M. (2005). Seasonal movement patterns, habitat use, and home range of flathead catfish in the Lower St. Joseph River, Michigan. *North American Journal of Fisheries Management* **25**, 256-269.

Declerck S., Vandekerkhove J., Johansson L., Muylaert K., Conde-Porcuna J.M., Van der Gucht K., Pérez-Martínez C., Lauridsen T., Schwenk K., Zwart G., Rommens W., López-Ramos J., Jeppesen E., Vyverman W., Brendonck L. & De Meester L. (2005). Multi-group biodiversity in shallow lakes along gradients of phosphorus and water plant cover. *Ecology* **86**, 1905-1915.

DeLeeuw J. & Mair P. (2009). Multidimensional scaling using majorization: SMACOF in R. *Journal of Statistical Software* **3**, 1-30.

DeLury D.B. (1951). On the Planning of Experiments for the Estimation of Fish Populations. *Journal of the Fisheries Research Board of Canada* **8**, 281-307.

Demirbas A. (2007). Focus on the world: Status and future of hydropower. *Energy Sources, Part B: Economics, Planning, and Policy* **2**, 237-242.

Denic M. & Geist J. (2010). Habitat suitability analysis for lacustrine brown trout (*Salmo trutta*) in Lake Walchensee: implications for the conservation of an endangered flagship species. *Aquatic Conservation: Marine and Freshwater Ecosystems* **20**, 9-17.

Deppe E. & Lathrop R.C. (1993). Recent changes in the aquatic macrophyte community of Lake Mendota. *Transactions of the Wisconsin Academy of Sciences, Arts and Letters* **81**, 47-58.

DIN 38410-1. (2004). *German standard methods for the examination of water, waste water and sludge - Biological-ecological analysis of water (group M) - Part 1: Determination of the saprobic index in running waters (M 1)*, Beuth Verlag, Berlin, Deutschland.

DIN EN 15204. (2006). *Water quality - Guidance standard on the enumeration of phytoplankton using inverted microscopy (Utermöhl technique)*, Beuth Verlag, Berlin, Deutschland.

Donaghy M.J. & Verspoor E. (2000). A new design of instream incubator for planting out and monitoring Atlantic salmon eggs. *North American Journal of Fisheries Management* **20**, 521-527.

Douglas M.M., Bunn S.E. & Davies P.M. (2005). River and wetland food webs in Australia's wet-dry tropics: general principles and implications for management. *Marine and Freshwater Research* **56**, 329-342.

Downs P.W. & Kondolf M.G. (2002). Post-project appraisals in adaptive management of river channel restoration. *Environmental Management* **29**, 477-496.

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

Dumas J. & Marty S. (2006). A new method to evaluate egg-to-fry survival in salmonids, trials with Atlantic salmon. *Journal of Fish Biology* **68**, 284-304.

Dußling U., Berg R., Klinger H. & Wolter C. (2004). *Assessing the Ecological Status of River Systems Using Fish Assemblages.* Handbuch Angewandte Limnologie, 20. Erg. Lfg. 12/04: 1-84. Auflage, Verison 8.0.6.

Dußling U., Bischoff A., Haberbosch R., Hoffmann A., Klinger H., Wolter C., Wysujack K. & Berg, R. (2005). The Index of Fish Regions (FRI) – a Tool for River Assessment According to the EC Water Framework Directive. *Wasserwirtschaft* **7-8**, 19-24.

Dynesius M. & Nilsson C. (1994). Fragmentation and flow regulation of river systems in the northern third of world. *Science* **266**, 753-762.

Dziock F., Henle K., Foeckler F., Follner K. & Scholz M. (2006). Biological Indicator Systems in Floodplains – a Review. *International Review of Hydrobiology* **4**, 271-291.

Ettl H., Gärtner G., Gerloff J., Heynig H. & Mollenhauer D. (1978-1999). *Süßwasserflora von Mitteleuropa*. Volumes 1-17&19. Gustav Fischer Verlag. Stuttgart, Germany.

European Parliament (2000). Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy. *Official Journal of the European Union* **327**, 1-73.

European Parliament (2000). Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for the Community action in the field of water policy. Annex V, Annex VI, Annex VII, Annex IX. *Official Journal of the European Union* **327**, 33-69.

European Parliament (2006). Directive 2006/11/EC of the European Parliament and of the Council on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community. *Official Journal of the European Union* **64**, 52-59.

European Parliament (2007). Directive 2007/60/EC of the European Parliament and of the Council on the assessment and management of flood risks. *Official Journal of the European Union* **288**, 27-35.

Erös T. (2007). Partitioning the diversity of riverine fish: the roles of habitat types and non-native species. *Freshwater Biology* **52**, 1400-1415.

Erös T., Botta-Dukat Z. & Grossmann G.D. (2003). Assemblage structure and habitat use of fishes in a central European submontane stream: a patch-based approach. *Ecology of Freshwater Fish* **12**, 141-150.

Erskine W.D., Terrazzolo N. & Warner R.F. (1999). River rehabilitation from the hydrogeomorphic impacts of a large hydro-electric power project: Snowy River, Australia. *Regulated Rivers: Research & Management* **15**, 3-24.

Fame Consortium (2004). *Manual for the application of the European Fish Index - EFI*. A fish-based method to assess the ecological status of European rivers in support of the Water Framework Directive. Version 1.1, January 2005.

Findlay S. (2010). Stream microbial ecology. *Journal of the North American Benthological Society* **29**, 170-181.

Fischer H., Pusch M. & Schwoerbel J. (1996). Spatial distribution and respiration of bacteria in stream-bed sediments. *Archiv für Hydrobiologie* **3**, 281-300.

Freeman M.C., Bowen Z.H., Bovee K.D. & Irwin E.R. (2001). Flow and habitat effecs on juvenile fish abundance in natural and altered flow regimes*. Ecological Applications* **11**, 179-190.

Fleeger J.W., Carmana K.R. & Nisbet R.M. (2003). Indirect effects of contaminants in aquatic ecosystems. *The Science of the Total Environment* **317**, 207-233.

Gayraud S. & Phillipe M. (2003). Influence of Bed-Sediment Features on the Interstitial Habitat Available for Macroinvertebrates in 15 French Streams. *International Review of Hydrobiology* **88**, 77-93.

Geist J. & Auerswald K. (2007). Physicochemical stream bed characteristics and recruitment of the freshwater pearl mussel (Margaritifera margaritifera). *Freshwater Biology* **55**, 2299-2316.

Geist J., Porkka M. & Kuehn R. (2006). The status of host fish populations and fish species richness in european freshwater pearl mussel (Margaritifera margaritifera L.) streams. *Aquatic Conservation: Marine and Freshwater Ecosystems* **16**, 251-266.

Geist J., Kolahsa M., Gum B. & Kuehn R. (2009). The importance of genetic cluster recognition for the conservation of migratory fish species: the example of the endangered European Huchen (*Hucho hucho* L.). *Journal of Fish Biology* **75**, 1063-1078.

Geist J. (2010). Strategies for the conservation of endangered freshwater pearl mussels (*Margaritifera margaritifera* L.): a synthesis of Conservation Genetics and Ecology. *Hydrobiologia* **644**, 69-88.

Geist J. (2011). Integrative freshwater ecology and biodiversity conservation. *Ecological Indicators* **11**, 1507-1516.

Giller P.S. (2005). River restoration: seeking ecological standards. Editor's introduction. *Journal of Applied Ecology* **42**, 201-207.

Gleick P.H. (1996). Water resources, in: Schneider S.H. (Ed.), *Encyclopedia of climate and weather*. Oxford University Press, New York, USA, pp. 817-823.

Gleick P.H. (2003). Global freshwater resources: soft-path solutions for the 21st century. *Science* **302**, 1524-1528.

Gowans A.R.D., Armstrong J.D., Priede I.G. & Mckelvey S. (2003). Movement of Atlantic Salmon migrating upstream through a fish-pass complex in Scotland. *Ecology of Freshwater Fish* **12**, 177-189.

Grossman G.D., De Sostoa A., Freemann M.C. & Lobon-Cervia J. (1987). Microhabitat use in a mediterranean riverine fish assemblage. Fishes of the lower Matarrana. *Oecologia* **73**, 490-500.

Grossman G.D. & De Sostoa A. (1994a). Microhabitat use by fish in the lower Rio Matarrana Spain, 1984–1987. *Ecology of Freshwater Fish* **3**, 123-136.

Grossman G.D. & De Sostoa A. (1994b). Microhabitat use by fish in the upper Rio Matarrana Spain, 1984–1987. *Ecology of Freshwater Fish* **3**, 141-152.

Grossman G.D. & Ratajczak R.E. (1998). Long-term patterns of microhabitat use by fish in a southern Appalachian stream from 1983 to 1992: effects of hydrologic period, season and fish length. *Ecology of Freshwater Fish* **7**, 108-131.

Grost R. T., Wayne A. H. & Wesche T. A. (1991). Description of brown trout redds in a mountain stream. *Transactions of the American Fisheries Society* **120**, 582-588.

Habit E., Belk M. & Parra O. (2007). Response of the riverine fish community to the construction and operation of a diversion hydropower plant in central Chile. *Aquatic Conservation: Marine and Freshwater Ecosystems* **17**, 37-49.

Hancock P.J., Boulton A.J. & Humphreys W.F. (2005). Aquifers and hyporheic zones: Towards an ecological understanding of groundwater. *Hydrogeological Journal* **13**, 98-111.

Harris G. S. (1973). A simple egg box planting technique for estimating the survival of eggs deposited in stream gravel. *Journal of Fish Biology* **5**, 85-88.

Harsányi A. (1982). *Der Huchen: Vorkommen, Aufzucht und sportl. Fang*. Hamburg und Berlin: Paul Parey Verlag, Deutscland.

Hauer C., Unfer G., Schmutz S. & Habersack H. (2008). Morphodynamic effects on the habitat of juvenile cyprinids (Chondrostoma nasus) in a restored Austrian lowland river. *Environmental Management* **42**, 279-296.

Heermann L. & Borcherding J. (2006). Winter short-distance migration of juvenile fish between two floodplain water bodies of the Lower River Rhine. *Ecology of Freshwater Fish* **15**, 161-168.

Heino J., Paavola R., Virtanen R. & Muotka T. (2005). Searching for biodiversity indicators in running waters: do bryophytes, macroinvertebrates, and fish show congruent diversity patterns? *Biodiversity and Conservation* **14**, 415-428.

Heino J. (2010). Are indicator groups and cross-taxon congruence useful for predicting biodiversity in aquatic ecosystems? *Ecological Indicators* **10**, 112-117.

Holčík J. (1990). Conservation of the huchen, *Hucho hucho* (L.), (Salmonidae) with special reference to Slovakian rivers. *Journal of Fish Biology* **37** (Suppl. A), 113-121.

Holčík J. (1995). Threatened fishes of the world: *Hucho hucho* (Linnaeus, 1758) (Salmonidae). *Environmental Biology of Fishes* **43**, 105-106.

Humphries P., Serafini L.G. & King A.J. (2002). River regulation and fish larvae: variation through space and time. *Freshwater Biology* **47**, 1307-1331.

Ihssen P.E., Booke H.E., Casselman J.M., McGlade J.M., Payne, N.R. & Utter F.M. (1981). Stock Identification: Materials and Methods. *Canadian Journal of Fisheries and Aquatic Sciences* **38**, 1838-1855.

Jenkins M. (2003). Prospects for Biodiversity. *Science* **302**, 1175-1177.

Jensen A.J. & Aass P. (1995). Migration of a fast-growing population of brown trout (*Salmo trutta* L.) through a fish ladder in relation to water flow and water temperature. *Regulated Rivers: Research & Management* **10**, 217-228.

John D.C., Whitton B.A. & Brook A.J. (2002). *The freshwater algal flora of the British Isles: An identification guide to freshwater and terrestrial algae*. University Press, Cambridge, United Kingdom.

Johnson R.K., Hering D., Furse M.T. & Clarke R.T. (2006). Detection of ecological change using multiple organism groups: metrics and uncertainty. *Hydrobiologia* **566**, 115-137.

Jungwirth M., Muhar S. & Schmutz S. (1995). The effects of recreated instream and ecotone structures on the fish fauna of an epipotamal river. *Hydrobiologia* **303**, 195-206.

Jungwirth M., Muhar S. & Schmutz S. (2000). Fundamentals of fish ecological integrity and their relation to the extended serial discontinuity concept. *Hydrobiologia* **422**/**423**, 85-97.

Jungwirth M., Haidvogl G., Moog O., Muhar S. & Schmutz S. (2003). *Angewandte Fischökologie an Fließgewässern*. 1st Edition, UBT Stuttgart, Germany.

Jurajda P. (1995). Effect of channelization and regulation on fish recruitment in a flood plain river. *Regulated Rivers: Research & Management* **10**, 207-215.

Jurajda P. (1999). Comparative nursery habitat use by 0+ fish in a modified lowland river. *Regulated Rivers: Research & Management* **15**, 113-124.

Karr J.R. (1981). Assessment of biotic integrity using fish communities. *Fisheries* **6**, 21-27.

Kemp P.S. & O'Hanley J.R. (2010). Procedures for evaluating and prioritising the removal of fish passage barriers: a synthesis. *Fisheries Management and Ecology* **17**, 297-322.

Kim J.G., Ko K.S., Kim T.H., Lee G.H., Chon C.M. & Lee J.S. (2007) Effects of mining and geology on the chemistry of stream water and sediment in a small watershed. *Geosciences Journal* **11**, 175-183.

Kirchhofer A. (1995). Morphological variability in the ecotone - an important factor for the conservation of fish species richness in Swiss rivers. *Hydrobiologia* **303**, 103-110.

Klupp R. (2000). *Fische und ihre Welt in Oberfranken*. Arbeitsgemeinschaft des Bezirks Oberfranken, des Bezirksfischereiverbandes Oberfranken e.V. und der Teichgenossenschaft Oberfranken, Bayreuth, Deutschland.

Knaepkens G., Baekelandt K. & Eens M. (2006). Fish pass effectiveness for bullhead (*Cottus gobio*), perch (*Perca fluviatilis*) and roach (*Rutilus rutilus*) in a regulated lowland river. *Ecology of Freshwater Fish* **15**, 20-29.

Knoben R.A.E., Roos C. & Van Oirschot M.C.M. (1995). *Biological Assessment Methods for Watercourses*. UN/ECE Task Force on Monitoring and Assessment, RIZA report nr. **95.066**, pp. 1-86.

Koleff P., Gaston,K.J. & Lennon J.J. (2003). Measuring beta diversity for presence-absence data. *Journal of Animal Ecology* **72**, 367-382.

Kondolf G.M., Anderson S., Lave R., Pagano L., Merenlender A. & Bernhardt E.S. (2007). Two decades of river restoration in California: what can we learn? *Restoration Ecology* **15**, 516-523.

Kottelat M. & Freyhof J. (2007). *Handbook of European freshwater fishes*. Kottelat, Cornol, Switzerland and Freyhof, Berlin, Germany.

Kruk A. (2007). Role of habitat degradation in determining fish distribution and abundance along the lowland Warta River, Poland. *Journal of Applied Ichthyology* **23**, 9-18.

Laffaille P., Acou A., Guillouet J. & Legault A. (2005). Temporal changes in European eel, *Anguilla anguilla,* stocks in a small catchment after installation of fish passes. *Fisheries Management and Ecology* **12**, 123-129.

Lenat D.R. (1988). Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. *Journal of the North American Benthological Society* **7**, 222-233.

Leopold A. (1948). *Sand County Almanac*. Oxford University Press, New York, USA.

Leuner E., Klein M., Bohl E., Jungblut J.H., Greber J. & Groh K. (2000). *Ergebnisse der Artenkartierung in den Fließgewässern Bayerns.* Bayerisches Staatsministerium für Ernährung, Landwirtschaft und Forsten, Presse- Druck- und Verlags- GmbH, Augsburg, Deutschland.

Lichatowich J. (1999). *Salmon without Rivers: A History of the Pacific Salmon Crisis*. New York: Island Press, New York USA.

Liess M. & Von der Ohe P.C. (2005). Analyzing effects of pesticides on invertebrate communities in streams. *Environmental Toxicology and Chemistry* **24**, 954-965.

Linke S., Turak E. & Nel J. (2011). Freshwater conservation planning: the case for systematic approaches. *Freshwater Biology* **56**, 6-20.

Lucas M.C. & Baras E. (2001). *Migration of Freshwater Fishes.* Blackwell Sciences Ldt.: London. pp. 66-87.

Lundberg U. (2004). Behavioural elements of the noble crayfish, Astacus astacus (Linnaeus, 1758). *Crustaceana* **77**, 137-162.

MacCrimmon H. R., Gots B. L. & Witzel L. D. (1989). Simple apparatus for assessing embryo survival and alevin emergence of stream salmonids. *Aquaculture and Fisheries Management* **20**, 333-335.

Mackenzie C. & Moring J. R. (1988). Estimating survival of Atlantic salmon during the intragravel period. *North American Journal of Fisheries Management* **8**, 45-49.

Makrakis S., Miranda L.E., Gomes L.C., Makrakis C.M. & Junior H.M.F. (2011). Ascent of neotropical migratory fish in the Itaipu reservoir fish pass. *River Research and Applications* **27**, 511-519.

Malakoff D. (2004). The river doctor. *Science* **305**, 937-939.

Marchetti M.P. & Moyle P. (2001). Effects of flow regime on fish assemblages in a regulated California stream. *Ecological Applications* **11**, 530-539.

Marchetti M.P., Light T., Moyle P. & Viers J.H. (2004). Fish invasions in California watersheds: testing hypothesis using landscape patterns. *Ecological Applications* **14**, 1507-1525.

McGeoch M.A. (1998) The selection, testing and application of terrestrial insects as bioindicators. *Biological Review.* **73**, 181-201.

Meier C., Haase P., Rolauffs P., Schindehütte K., Schöll F., Sundermann A. & Hering D. (2006). *Methodisches Handbuch Fließgewässerbewertung. Handbuch zur Untersuchung und Bewertung von Fließgewässern auf der Basis des Makrozoobenthos vor dem Hintergrund der EGWasserrahmenrichtlinie.* http://www.fliessgewaesserbewertung.de/downloads/abschlussbericht_20060331_anhang_IX.pdf.

Menezes S., Baird D.J. & Soares A.M.V.M. (2010). Beyond taxonomy: a review of macroinvertebrate trait-based community descriptors as tools for freshwater biomonitoring. *Journal of Applied Ecology* **47**, 711-719.

Merrit R. W. & Cummins K. W. (1996). *An Introduction of the aquatic insects of North America*, Kendall/Hunt, Dubuque, IA, USA.

Milan D.J., Petts G.E. & Sambrook H. (2000). Regional variations in the sediment structure of trout streams in southern England: benchmark data for siltation assessment and restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems* **10**, 407-420.

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-Being: Current State and Trends*. Island Press, Washington, DC, USA.

Minns C.K., Kelso J.R.M. & Randall R.G. (1996). Detecting the response of fish to habitat alterations in freshwater ecosystems. *Canadian journal of Fisheries and Aquatic Sciences* **53**, 403-414.

Moog O. (1995). *Fauna Aquatica Austriaca*. 1st. Edition, Wasserwirtschaftskataster, Federal Ministry for Agriculture and Forestry, Vienna.

Mueller M., Pander J. & Geist J. (2011). The effects of weirs on structural stream habitat and biological communities. *Journal of Applied Ecology* **48**, 1450-1461.

Müllner A.N. & Schagerl M. (2003). Abundance and vertical distribution of the phytobenthic community within a pool and riffle sequence of an Alpine gravel stream. *International Review of Hydrobiology* **88**, 243-254.

Myers N. (1988). Threatened biotas: 'hot-spots' in tropical forests. *Environmentalist* **8**, 187-208.

Mykrä H., Heino J. & Muotka T. (2007). Scale related patterns in the spatial and environmental components of stream macroinvertebrate assemblage variation. *Global Ecology and Biogeography* **16**, 149-159.

Nagel P., Bäthe J., Günther C., Heinemann R., Kunz M., Potel S., Schönleber Y., Steinhausen S. & Trischler M. (1989). *Bildbestimmungsschlüssel der Saprobien*. Gustav Fischer Verlag Stuttgart, New York, USA.

Naiman R.J. (2008). Freshwater animal diversity assessment - foreword. *Hydrobiologia* **595**, 1-2.

Neumann M., Liess M. & Schulz R. (2003). An expert system to estimate the pesticide contamination of small streams using benthic macroinvertebrates as bioindicators, Part 2: The knowledge base of LIMPACT. *Ecological Indicators* **2**, 391-401.

Nienhuis P.H. & Gulati R.D. (2002). The state of the art of aquatic and semi-aquatic ecological restoration projects in the Netherlands. *Hydrobiologia* **478**, 219-233.

Nilsson C., Reidy C.A., Dynesius M. & Revenga, C. (2005). Fragmentation and flow regulation of the world's large river systems. *Science* **308**, 405-408.

Noss R.F., LaRoe E.T. & Scott J. (1995). *Endangered ecosystems of the United States: a preliminary assessment of loss and degradation.* U.S. Dept. of the Interior, National Biological Service, Washington, D.C., USA.

Nykänen M., Huusko A. & Mäki-Petäys A. (2001). Seasonal changes in the habitat use and movements of adult European grayling in a large subarctic river. *Journal of Fish Biology* **58**, 506-519.

Oksanen J., Kindt R., Legendre P., O'Hara B., Simpson G.L., Solymos P., Stevens M.H.H. & Wagner H. (2009). vegan: Community Ecology Package. R package version 1.15-3.

Paavola R., Muotka T., Virtanen R., Heino J., Jackson D. & Mäki-Petäys A. (2006). Spatial Scale Affects Community Concordance Among Fishes, Benthic Macroinvertebrates, And Bryophytes In Streams. *Ecological Applications* **16**, 368-379.

Palmer M. A., Covich A. P., Finlay B. J., Gibert J., Hyde K. D., Johnson R. K., Kairesalo T., Lake S., Lovelle C. R., Naiman R. J., Ricci C., Sabater F. & Strayer D. (1997). Biodiversity and ecosystem processes in freshwater sediments. *Ambio* **26**, 571-577.

Palmer M.A., Bernhardt E.S., Chornesky E., Collins S., Dobson A., Duke C., Gold B., Jacobson R., Kingsland S., Kranz R., Mappin M., Martinez M.L., Micheli F., Morse J., Pace M., Pascual M., Palumbi S., Reichmann O.J., Simons A., Townsend A. & Turner M. (2004). Ecology for a Crowded Planet. *Science* **304**, 1251-1252.

Palmer M. A., Bernhardt E. S., Allan J. D., Lake P. S., Alexander G., Brooks S., Carr J., Clayton S., Dahm C. N., Follstad Shah J., Galat D. L., Loss S. G., Goodwin P., Hart D. D., Hassett B., Jenkinson R., Kondolf G. M., Lave R., Meyer J. L., O'Donnell T. K., Pagano L. & Sudduth E. (2005). Standards for ecologically successful river restoration. *Journal of Applied Ecology* **42**, 208-217.

Pander J., Schnell J., Sternecker K. & Geist J. (2009). The "egg sandwich": a method for linking spatially resolved salmonid hatching rates with habitat variables in stream ecosystems. *Journal of Fish Biology* **74**, 683-690.

Pander J. & Geist J. (2010a). Seasonal and spatial bank habitat use by fish in highly altered rivers, a comparison of four different restoration measures. *Ecology of Freshwater Fish* **19**, 127-138.

Pander J. & Geist J. (2010b). Salmonid-egg floating boxes as bioindication for riverine water quality and stocking success. *Journal of Fish Biology* **76**, 2584-2590.

Pander J., Mueller M. & Geist J. (2011). Ecological functions on fish bypass channels in streams: migration corridor and habitat for rheophilic species. *River Research and Applications*, online early DOI: 10.1002/rra.1612.

Pauwels S. J. & Haines T. A. (1994). Survival, hatching, and emergence success of Atlantic salmon eggs planted in three Maine streams. *North American Journal of Fisheries Management* **14**, 125-130.

Pelicice F., Agostinho A.A. (2008). Fish-Passage Facilities as Ecological Traps in Large Neotropical Rivers. *Conservation Biology* **22**, 180-188.

Piegy H. & Gurnell A.M. (1997). Large woody debris and river geomorphological pattern: examples from S.E. France and S. England. *Geomorphology* **19**, 99-116.

Pielou E.C. (1966). The measurement of diversity in different types of biological collections. *Journal of Theoretical Biology* **13**, 131-144.

Pimm S.L., Ayres M., Balmford A., Branch G., Brandon K., Brooks T., Bustamante R., Costanza R., Cowling R., Curran L.M., Dobson A., Farber S., Fonseca da G.A.B., Gascon C., Kitching R., McNeely J., Lovejoy T., Mittermeier R.A., Myers N., Patz J.A., Raffle B., Rapport D., Raven P., Roberts C., Rodriguez J.P., Rylands A.B., Tucker C., Safina C., Samper C., Stiassny M.L.J., Supriatna J., Wall D.H. & Wilcove D. (2001). Can we defy nature's end? *Science* **293**, 2207-2208.

Poff N.L. & Ward J.V. (1990). Phyisical habitat template of lotic systems: recovery in the context of historic pattern of spatial heterogeneity. *Environmental Management* **14**, 629-645.

Power M. (1999). Recovery in aquatic ecosystems: an overview of knowledge and needs. *Journal of Aquatic Ecosystem Stress and Recovery* **6**, 253-257.

Prenda J., Armitage P.D. & Grayston A. (1997). Habitat use by fish assemblages of two chalk streams. *Journal of Fish Biology* **51**, 64-79.

Pyšek P. & Richardson D.M. (2010). Invasive Species, Environmental Change and Management, and Health. *Annual Review of Environment and Resources* **35**, 25-55.

Quist M.C., Rahel F.J. & Hubert W.A. (2005). Hierarchical faunal filters: an approach to assessing effects of habitat and nonnative species on native fishes. *Ecology of Freshwater Fish* **14**, 24-39.

R Development Core Team (2008). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Rawer-Jost C., Böhmer J., Blank J. & Rahmann H. (2000). Macroinvertebrate funcitonal feeding group methods in ecological assessment. *Hydrobiologia* **422/423**, 225-232.

Reid W.V. (1997). Strategies for conserving biodiversity. *Environment* **39**, 16-43.

Renöfält B.M., Jansson R. & Nilsson C. (2010). Effects of hydropower generation and opportunities for environmental flow management in Swedish riverine ecosystems. *Freshwater Biology* **55**, 49-67.

Revenga C., Campbell I., Abell R., de Villiers P. & Bryer M. (2005). Prospects for monitoring freshwater ecosystems towards the 2010 targets. *Philosophical Transactions of the Royal Society B* **360**, 397-413.

Ricciardi A. & Rasmussen B. (1999). Extinction rates of North American freshwater fauna. *Conservation Biology* **13**, 1220-1222.

Rice S. P., Lancaster J. & Kemp P. (2010). Experimentation at the interface of fluvial geomorphology, stream ecology and hydraulic engineering and the development of an effective interdisciplinary river science. *Earth Surface Processes and Landforms* **35**, 64-77.

Richards C., Haro R.J., Johnson L.B., Host G.E. (1996). Landscape-scale influences on stream habitats and biota. *Canadian Journal of Fisheries and Aquatic Sciences* **53**, 295-311.

Rivinoja P., McKinnell S. & Lundqvist H. (2001). Hindrances to upstream migration of atlantic salmon (*Salmo salar*) in a northern Swedish river caused by a hydroelectric power station. *Regulated Rivers: Research and Management* **17**, 101-115.

Robinson H.W. (1998). Influence of drainage connectivity, drainage area and regional species richness on fishes of the interior highlands in Arkansas. *American Midland Nature* **139**, 1-19.

Rolauffs P., Hering D., Sommerhäuser M., Rödiger S. & Jähnig S. (2003). Entwicklung eines leitbildorientierten Saprobienindexes für die biologische Fließgewässerbewertung. *Umweltbundesamt Texte* **11/03**, 1-137.

Roscoe D.W. & Hinch S.G. (2010). Effectiveness monitoring of fish passage facilities: historical trends, geographic patterns and future directions. *Fish and Fisheries* **11**, 12-33.

Rosenberg D.M. (2000). Global-scale enviromental effects of hydrological alterations. *BioScience* **50**, 746-51.

Rosenberg D.M., Berkes F., Bodaly R.A., Hecky R.E., Kelly C.A. & Rudd J.W.M. (1997). Large-scale impacts of hydroelectric development. *Environmental Reviews* **5**, 27-54.

Rubin J.-F. (1995). Estimating the success of natural spawning of salmonids in streams. *Journal of Fish Biology* **46**, 603-622.

Rubin J.-F. & Glimsäter C. (1996). Egg-to-fry survival of the sea trout in some streams of Gotland. *Journal of Fish Biology* **48**, 585-606.

Santos J.M., Ferreira M.T., Pinheiro A.N. & Bochechas J.H. (2006). Effects of small hydropower plants on fish assemblages in medium-sized streams in central and northern Portugal. *Aquatic Conservation: Marine and Freshwater Ecosystems* **16**, 373-388.

Schaefer J.F. & Kerfoot J.R. (2003). Fish assemblage dynamics in an adventitious stream: a landscape perspective. *American Midland Nature* **151**, 134-145.

Schaumburg J., Schmedtje U., Köpf B., Schranz C., Schneider S., Stelzer D. & Hofmann G. (2006). *Handlungsanweisung für die ökologische Bewertung von Seen zur Umsetzung der*

EU-Wasserrahmenrichtlinie: Makrophyten und Phytobenthos - Stand September 2006. Bayerisches Landesamt für Umwelt, Deutschland.

Schiemer F., Keckeis H. & Kamler E. (2003). The early life history stages of riverine fish: ecophysiological and environmental bottelnecks. *Comparative Biochemistry and Physiology Part A* **133**, 439-449.

Schmedtje U. & Colling M. (1996). Ökologische Typisierung der aquatischen Makrofauna. *Informationsberichte des Bayerischen Landesamtes für Wasserwirtschaft* **4/96**, Deutschland.

Schmidt H., Bernet D., Wahli T., Meier W. & Burkhardt-Holm P. (1999). Active biomonitoring with brown trout and rainbow trout in diluted sewage plant effluents. *Journal of Fish Biology* **54**, 585-596.

Schneider S. & Melzer A. (2003). The trophic index of macrophytes (TIM) – a new tool for indicating the trophic state of running waters. *International Review of Hydrobiology* **88**, 49-67.

Schneider S. & Lindstrøm E.-A. (2009). Bioindication in Norwegian rivers using non-diatomaceous benthic algae: the acidification index periphyton (AIP). *Ecological Indicators* **9**, 1206-1211.

Schneider S. & Lindstrøm E.-A. (2011). The periphyton index of trophic status PIT: a new eutrophication metric based on non-diatomaceous benthic algae in Nordic rivers. *Hydrobiologia* **665**, 143-155.

Schubert R. (1991). *Bioindikation in terrestrischen Ökosystemen*. Gustav Fischer Verlag. Jena, Deutschland.

Sear D.A. (1994). River restoration and geomorphology. *Aquatic Conservation: Marine and Freshwater Ecosystems* **4**, 169-177.

Shannon C.E. & Weaver W. (1949). *The mathematical theory of communication*, Vol. 1. Urbana: University of Illinois Press, USA.

Simberloff D. (1998). Flagships, Umbrellas, and Keystones: is single-species management passé in the landscape era? *Biological Conservation* **83**, 247-257.

Sinowski W. & Auerswald K. (1999). Using relief parameters in a discriminant analysis to stratify geological areas with different spatial variability of soil properties. *Geoderma* **89**, 113-128.

Soininen J. (2002). Responses of epilithic diatom communities to environmental gradients in some Finnish rivers. *International Review of Hydrobiology* **87**, 11-24.

Soulsby C., Malcolm I.A. & Youngson A.F. (2001a). Hydrochemistry of the hyporheic zone in salmon spawning gravels: a preliminary assessment in a degraded agricultural stream. *Regulated Rivers: Reasearch & Management* **17**, 651-665.

Soulsby C., Youngson A.F., Moir,H.J. & Malcom I.A. (2001b). Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: a preliminary assessment. *The Science of Total Environment* **265**, 295-307.

Sparks R.E., Bayley P.B., Kohler S.L. & Osborne L.L. (1990). Disturbance and recovery of large floodplain rivers. *Environmental Management* **14**, 699-709.

Stanford J., Ward J.V., Liss W.J., Frissell C.A., Williams R.N., Lichatowich J.A. & Coutant C. (1996). A generalprotocol for restoration of regulated rivers. *Regulated Rivers: Research & Management* **12**, 391-413.

Stein B. A. & Flack S. R. (1997). *Species Report Card: The State of U.S. Plant and Animals*. Arlington, VA: The Nature Conservancy, USA.

Stendera S. & Johnson R.K. (2006). Multiscale drivers of water chemistry of boreal lakes and streams. *Environmental Management* **38**, 760-770.

Stevenson R.J., Pan Y., Manoylov K.M., Parker C.A., Larsen D.P. & Herilhy A.T. (2008). Development of diatom indicators of ecological conditions for streams of the western US. *Journal of the North American Benthological Society* **27**, 1000-1016.

Strayer D.L. & Dudgeon D. (2010). Freshwater biodiversity conservation: recent progress and future challenges. *Journal of the North American Benthological Society*. **29**, 344-358.

Surber E. W. (1930). A quantitative method of studying the food of small fishes. *Transactions of the American Fisheries Society* **60**, 158-163.

Szabo K.E., Makk J., Kiss K.T., Eiler A., Acs E., Tot, B. & Bertilsson S. (2008). Sequential colonization by river periphyton analysed by microscopy and molecular fingerprinting. *Freshwater Biology* **53**, 1359-1371.

Tarkan AS. (2006). Reproductive ecology of two cyprinid fishes in an oligotrophic lake near the southern limits of their distribution range. *Ecology of Freshwater Fish* **15**, 131-138.

United Nations (1992). *Convention on Biological* Diversity (text with annexes). United Nations, New York, USA.

Utermöhl H. (1931). Neue Wege in der quantitativen Erfassung des Planktons (mit besonderer Berücksichtigung des Ultraplanktons). *Internationale Vereinigung für theoretische und angewandte Limnologie* **5**, 567-596.

Van de Weyer K. & Schmidt C. (2007). *Bestimmungsschlüssel für die aquatischen Makrophyten (Gefäßpflanzen, Armleuchteralgen und Moose) in Deutschland*. Nettetal, lanaplan, Deutschland.

Vannote R.L., Minshall G.W., Cummins K.W., Sedell J.R. & Cushing C.E. (1980). The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences* **37**, 130-137.

Vibert R. (1949). Du repeuplement en truites et saumons par enfouissement de 'boites d' alevinage' garnies d' oeufs dans les graviers. *Bulletin Francxais de Pisciculture* **153**, 125-150.

Vinyoles D., De Sostoa A., Franch C., Maceda-Veiga A., Casals F. & Caiola N. (2010). Life-history traits of the stone loach *Barbatula barbatula*. *Journal of Fish Biology* **77**, 20-32.

Vlach P., Dusek J., Svatora M. & Moravec P. (2005). Fish assemblage structure, habitat and microhabitat preference of five fish species in a small stream. *Folia Zoologica* **54**, 421-431.

Vollestad L.A. & L'Abee-Lund J.H. (1987). Reproductive biology of stream-spawning roach, *Rutilus rutilus*. *Environmental Biology of Fishes* **18**, 219-227.

Ward J.V., Stanford J.A. (1983). *The serial discontinuity concept of lotic ecosystems.* Science Publicatons , Ann Arbor Mich., USA, pp. 29-42

Ward J.V., Tockner K., Uehlinger U. & Malard F. (2001). Understanding natural patterns and processes in river corridors as the basis for effective river restoration. *Regulated Rivers: Research & Management* **17**,311-323.

Wesche T.A. (1985). *Stream channel modification and reclamation structures to enhance fish habitat.* In: Gore, J.A., ed. The restoration of rivers and streams theories and experience, Chapter 5. Butterworth, Stoneham, MA, United Kingdom, pp. 1-64.

Whitlock D. (1979). *The Whitlock-Vibert Box Handbook*. El Segundo, CA: Federation of Fly Fishers, USA.

Wilson M.V. & Shmida A. (1984). Measuring beta diversity with presence-absence data. *Journal of Ecology* **72**, 1055-1064.

Winemiller K.O., Flecker A.S. & Hoeinghaus D.J. (2010). Patch dynamics and environmental heterogeneity in lotic ecosystems. *Journal of the North American Benthological Society* **29**, 84-99.

Wu J. & Skelton-Groth K. (2002). Targeting conservation efforts in the presence of threshold effects and ecosystem linkages. *Ecological Economics* **42**, 313-331.

Yu S. & Peters E.J. (2003). Diel and seasonal abundance of fishes in the Platte River, Nebraska, USA. *Fisheries Science* **69**, 154-160.

Zauner G. & Eberstaller J. (1999). Klassifizierungsschema der österreichischen Flußfischfauna in bezug auf deren Lebensraumansprüche. *Österreichs Fischerei* **52**, 198-205.

Zelinka M. & Marvan P. (1961). Zur Präzisierung der biologischen Klassifikation der Reinheit fliessender Gewässer. *Archives of Hydrobiology* **57**, 389-407.

Zhou S., Tang T., Wu N., Fu X. & Cai Q. (2008). Impacts of a small dam on riverine zooplankton. *International Review of Hydrobiology* **93**, 297-311.