River rehabilitation and fish
the challenge of initiating ecological recovery

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River rehabilitation and fish: The challenge of initiating ecological recovery

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SUMMARY

Today, the majority of the world's rivers and streams is strongly exploited by man. These exploitations have led to considerable degradation of running water ecosystems, as demonstrated by the threat to many riverine fish species or the extensive decline in wetlands. Ecosystem services, such as the provision of drinking water resources or flood retention, are also considerably affected by the degradation of river systems.

In recent years, increasing efforts have been undertaken to bring the gradual degradation to a halt and to return impaired river systems to a more natural state (restoration and rehabilitation, respectively). Often, an emphasis is placed on the recreation of the natural channel structure, which is expected to positively affect aquatic organisms. However, there was no obvious biological response in several projects. In this context, science has the exciting task of identifying the responsible recovery processes and contributing to an improved rehabilitation practice.

In this thesis, different stages of the rehabilitation process were studied in greater detail. On the one hand, surveys were conducted at the river Rhone in southern Switzerland, which is due to undergo large-scale rehabilitation as part of a flood protection project. On the other hand, the river Thur in north-eastern Switzerland served as a study system, in which rehabilitation began already in the early 1990ies. The scientific focus of this thesis lay primarily on fish ecology. Fish are highly suitable organisms for the detection of stream impairment and recovery due to their high mobility and longevity and the extensive knowledge available on their ecological role.

Only two out of 19 historically documented fish species were found in a system-wide survey of the river Rhone, which today is mainly canalised and intensively used for hydropower production. The analysis of historical topographic maps revealed the loss of 164 km (40 %) of shoreline habitats since 1850. These alterations are associated with the two systematic river corrections, which led to the isolation of the formerly vast floodplains. Today, 97 % of the Rhone plain exhibit an ecological deficit when compared with the situation around 1900.

An increase in habitat diversity was found in the largest river widenings at the river Thur relative to the canalised sections. In terms of the fish fauna, however, with our methodological approach no difference was detected between canalised and rehabilitated reaches, except for the locally increased winter densities in the widenings' well-structured backwaters. The current composition of the fish fauna deviated clearly from that described in a historical fish inventory.

In a literature review, 35 international case studies were investigated which report on the fish biological effectiveness of habitat rehabilitation projects in running waters. The analysis showed that the selection of evaluation parameters and reference sites had an important influence on the investigators' conclusion on project success. Based on the results of the review, guidelines for future river rehabilitation and monitoring projects were formulated.

The results gained in the present thesis demonstrate the particular significance of near-natural reference systems. In the planning phase prior to rehabilitation, information from near-natural references enables the assessment of a system's actual degradation and the development of adequate rehabilitation measures. Furthermore, historical maps can be used for the spatially explicit identification of priority sites for rehabilitation. After rehabilitation, a comparison with the near-natural reference conditions supplies important information on project effectiveness, i.e. on the naturalness of a rehabilitated site.

Rehabilitation ecology focuses increasingly on the reestablishment of ecosystem functions. These are, for instance, a natural flow or sediment regime, the migration of fish or the decomposition of organic matter. To evaluate whether ecosystem functions were improved, appropriate metrics, i.e. functional indicators, are needed. Functional indicators have been rarely used in fish biological project evaluations, as the literature analysis in this thesis revealed. The limited use of functional indicators may be attributed to their high survey costs and the lack of suitable guidelines.

In intensively used landscapes, running waters usually face multiple impacts. The studies on the rivers Rhone and Thur demonstrated clearly the need for large-scale and long-term measures to bring the expected success to rehabilitation projects.

ZUSAMMENFASSUNG

Die Mehrheit der Fliessgewässer weltweit ist stark durch den Menschen genutzt. Dies wirkt sich vielerorts negativ auf die aquatischen Lebensgemeinschaften aus, wie die Gefährdung zahlreicher Flussfischarten oder der massive Rückgang an Feuchtgebieten zeigen. Aber auch menschliche Güter sind von der Beeinträchtigung der Gewässer betroffen, so etwa die Versorgung mit Trinkwasser oder der Hochwasserrückhalt.

Mit der Revitalisierung von Fliessgewässern wird seit einigen Jahren versucht, der fortschreitenden Verschlechterung Einhalt zu gebieten und Flüsse und Bäche wieder in einen naturnäheren Zustand zurückzuführen. Häufig steht dabei die Wiederherstellung einer naturnahen Gewässerstruktur im Vordergrund, von der man sich eine positive Wirkung auf die aquatischen Lebewesen erhofft. In zahlreichen Projekten ist diese biologische Reaktion aber ausgeblieben. Es stellt sich der Wissenschaft die spannende Aufgabe, die verantwortlichen Prozesse zu identifizieren und damit zu einer verbesserten Revitalisierungspraxis beizutragen.

In der vorliegenden Arbeit wurden einzelne Stadien im Ablauf eines Revitalisierungsprojekts genauer untersucht. Als Untersuchungsgewässer diente einerseits die Rhone im Kanton Wallis, Schweiz, die vor einer grossräumigen Aufwertung im Rahmen eines Hochwasserschutzprojektes steht. Zum anderen fanden Aufnahmen an der Thur (Kantone Zürich und Thurgau, Schweiz) statt, die seit Beginn der 1990er Jahren über mehrere Strecken revitalisiert wurde. In den Untersuchungen lag der fachliche Schwerpunkt mehrheitlich bei der Fischfauna. Fische sind sehr geeignete Organismen, um die Lebensbedingungen in Fliessgewässern zu studieren, da sie relativ langlebig und mobil sind, und ihre ökologischen Ansprüche zudem meist gut dokumentiert sind.

Im Systemüberblick an der weitgehend kanalisierten und für die Stromproduktion stark genutzten Rhone wurden nur zwei der 19 historisch vorkommenden Fischarten nachgewiesen. Die Analyse historischer Kartenwerke zeigte zudem, dass seit 1850 im Rahmen der zwei systematischen Flusskorrekturen 164 km wertvoller Uferhabitate verloren gingen (40 %), insbesondere durch die Abkopplung ausgedehnter Schwemmflächen. Des Weiteren besteht für 97 % der Rhoneebene ein ökologisches Defizit, wenn mit der Situation um 1900 verglichen wird.

In den längsten Flussaufweitungen an der Thur findet sich eine gegenüber den kanalisierten Strecken erhöhte Habitatsdiversität. Bezüglich Fischfauna ergibt sich jedoch kein Unterschied zwischen kanalisierten und aufgewerteten Abschnitten, abgesehen von den lokal erhöhten Winterdichten in gut strukturierten Hinterwassern der Aufweitungen. In beiden Streckentypen weicht die heutige Zusammensetzung der Fischfauna deutlich von der historisch dokumentierten ab.

In einer Literaturanalyse wurden 35 internationale Fallstudien zu Habitatsaufwertungen mit fischökologischer Erfolgskontrolle ausgewertet. Diese zeigte, dass die Wahl der Untersuchungsgrössen sowie der Vergleichsstrecken die Aussage über den Projekterfolg entscheidend beeinflusst.

Die Resultate der vorliegenden Arbeit unterstreichen, wie wertvoll die Kenntnis von naturnahen Flusssystemen im Revitalisierungsprozess ist. Einerseits ermöglichen derartige naturnahe Referenzen, dass in der Planungsphase der eigentliche Grad der Beeinträchtigung bestimmt und passende Massnahmen entwickelt werden können. Ebenso wird eine Einschätzung möglich, an welchen Orten Revitalisierungen besonders dringlich sind. Andererseits bilden naturnahe Referenzen einen wichtigen Bestandteil der Erfolgskontrolle, indem der Natürlichkeitsgrad der revitalisierten Strecken eingeschätzt werden kann.

In der Revitalisierungsökologie wird verstärkt die Wiederherstellung von Ökosystem-Funktionen gefordert. Ökosystem-Funktionen sind beispielsweise ein natürliches Abfluss- oder Geschieberegime, die Wanderung von Fischen oder der Abbau von organischer Substanz. Damit das Erreichen dieser Zielsetzungen auch überprüft werden kann, sind entsprechende Messgrössen nötig, so genannte funktionelle Indikatoren.

Diese werden jedoch in den fisch-ökologischen Erfolgskontrollen bisher kaum eingesetzt, wie die Literaturanalyse in dieser Arbeit zeigte. Die begrenzte Verwendung funktioneller Indikatoren ist wohl auf die teils noch hohen Aufnahmekosten sowie die fehlenden Erhebungsanleitungen zurückzuführen. Auf diesem Gebiet besteht weiterer Forschungsbedarf.

In einer vielseitig genutzten Kulturlandschaft wie der Schweiz sind Fliessgewässer oft mehreren unterschiedlichen Beeinträchtigungen ausgesetzt. Die Studien an Rhone und Thur zeigen deutlich, dass grossräumige und langfristige Massnahmen nötig sind, um Revitalisierungsprojekte zum angestrebten Erfolg zu bringen.

CHAPTER 1

General Introduction

„The river Rhone is a monster that has to be domesticated."

Historical source, 18^{th} century

The history of human civilisation is closely related to the use of water (Gleick 2001). Traditionally, rivers have provided fishery resources, waterways for navigation, and water supply for agriculture, industry and domestic life. But despite these benefits and services (Ehrenfeld 2000), rivers have also posed threats to the local population (Petts 1989). They have hampered land use and fuelled water-related diseases such as malaria (Vischer 1989) by causing floods and channel migration over vast flood-plains, and creating extensive wetlands. Early river engineering was therefore a struggle against the river (Nienhuis and Leuven 2001) in order to reclaim land and to protect the growing infrastructure.

Over the last 200 to 500 years, human impact on European rivers has been greatly intensified (Nienhuis and others 2002b; Petts 1989). By the 19^{th} century, technological innovation allowed large-scale regulation even of highly dynamic, piedmont rivers (Petts 1989). This enabled economic revival also in formerly river-dominated regions, such as the valleys of the Swiss Rhone or the Alpine Rhine river. With the Industrial Revolution and the fast-growing population, the demand for water and energy grew further. In the early 20^{th} century, the construction of dams and reservoirs increased, with a peak occurring between 1950 and 1970 (Petts 1989). Hydropower has become an extremely important energy source providing nearly one fifth of the world-wide electricity (Gleick 2001). In Switzerland, hydropower makes up 58 % of the today's domestic energy production (Truffer and others 2001).

Human activities have shown detrimental impacts on riverine ecosystems. Sewage and industrial wastes or chemical spills have led to serious pollution of entire river courses, such as the British Thames (Gameson and Wheeler 1977) or the River Rhine (Capel and others 1988). Canalised rivers are often isolated from their floodplains and display monotonous channel morphology (Ward and others 1999). Dams disrupt the river continuum (Vannote and others 1980), altering important ecological processes, such as sediment and flow regime or the dispersal of organisms (Lytle and Poff 2003; Poff and Hart 2002). Furthermore, the river stretch between water diversion and release often has only a reduced discharge, i.e. residual flow. Reservoir operation may lead to short-term flow fluctuations or hydropeaking (Heggenes 1988).

Expressed in figures, these impacts read as follows: Today, 59 % of the world's large river systems are moderately to strongly affected by flow regulation, such as reservoir operation, irrigation or fragmentation by dams (Nilsson and others 2005). In the United States only 2 % of rivers and streams are categorised as relatively natural and more than one third is considered as heavily impaired or polluted (Benke 1990). Today, the majority of lowland rivers in Europe is canalised (Brookes and Shields 1996). In North America and Europe, 90 % of all floodplains have lost their ecological functioning due to cultivation (Tockner and Stanford 2002). About 80 % (3'900 km) of the larger Austrian river courses are classed as moderately to heavily impacted by human activity (Muhar and others 2000). More than 38 % (23'000 km) of the total Swiss river network are little to strongly affected by morphological impairments. Another 24 % (14'600 km) is in an artificial state or flows in culverts (Peter 2006; Peter and others 2005). About 25 % of the larger hydropower plants in Switzerland (> 300 kWh per year) produce hydropeaks thereby affecting approximately 30 % of all the country's rivers and streams (Limnex AG 2001).

Different impacts often occur simultaneously leading to heavily impaired river systems. The Swiss river Rhone, for instance, is mainly canalised due to two corrections carried out in the late 19^{th} and early 20^{th} centuries. As a result of hydrological changes arising from the exploitation of hydropower in the catchment (Loizeau and Dominik 2000) reaches with a near-natural flow regime are rare. The mean natural annual discharge has been reduced by more than 20 % along 22 % (36 km) of the entire river course between the river's source and Lake Geneva. Hydropeaking prevails over a distance of 109 km (65 %; data from Spreafico and others 1992), leading to significantly increased winter flows (Meile and others 2005).

Owing to multiple impacts, running waters are among the most threatened ecosystems worldwide (Dudgeon and others 2006). Projected extinction rates of North American freshwater fauna exceed those for terrestrial biota by a factor of five (Ricciardi and Rasmussen 1999). Of those fish species considered in the 2000 IUCN Red List, approximately 30 % (mostly freshwater) are threatened (Arthington and others 2004). Today, only 12 (26 %) of the 46 native species still occurring in Switzerland are classed as not threatened (VBGF). Eight species have already become extinct.

Ecosystem services are also considerably affected by the degradation of running water systems. Drinking water resources are threatened by impaired water quality, reduced capacity for self-purification of the remaining floodplains and lowering of the groundwater table. Flood retention is greatly reduced as a consequence of the diminished floodplain area (Nienhuis and others 2002a). In combination with the early canalisation work, the risk of extremely high water peaks has increased. This tendency is predicted to rise further as a consequence of higher precipitation associated with global warming and intensive land use in the catchments. Fisheries resources and fish catches in rivers have significantly declined in many parts of the world (Burkhardt-Holm and others 2002; Cowx and others 2004). Additionally, rivers of low naturalness are usually of reduced recreational value.

„Within the realms of possibility, the main goal is to give more space to the river Rhone, in order to increase diversity, to restore ecological functions of the fluvial system and to recreate alluvial dynamics."

Dienststelle für Strassen- und Flussbau des Kanton Wallis (2000)

Recognition of all these impairments and catastrophic events, such as the Sandoz accident on the river Rhine in 1986 led to a paradigm shift in river regulation that Nienhuis and others (2002b) call the change from technological to ecological river management. The need for sustainability and integral management of water resources has been increasingly acknowledged and, today, ecological, social and economical concerns are equally highlighted in the legislation of many countries, such as the European Water Framework Directive (WFD) or the Swiss Federal Constitution. The WFD aims to achieve a good ecological status for all aquatic ecosystems, i.e. inland surface waters, transitional waters, coastal waters and groundwater, by 2015. Specific objectives of the WFD are the protection of these systems from further deterioration, the enhancement of their ecological status, the promotion of a sustainable use of water resources and the diminution of the ecological effects of floods and droughts. The implementation of the directive is carried out in accordance with specific management plans that are elaborated at the scale of river basin districts.

The restoration of degraded rivers is an important element of these efforts. Compared to the 1970ies and 80ies, emphasis is no longer predominantly on the chemical water quality. In many European countries, water quality has been significantly improved over the last 30 years (Nienhuis and others 2002b), primarily as a consequence of the introduction of water treatment stations and legal regulations. Today, river restoration is mainly focused on the remediation of the physical habitat.

„Rehabilitating river habitats to enhance biodiversity recovery"

Aarts and others (2004)

In a river system that has been modified by human use, essential natural functions or processes may be altered or even disrupted (Angermeier 1997; Bradshaw 1997). Such functions and processes are, e.g., delivery of wood, cycling of organic matter, disturbance regime or migration and reproduction of a particular species. This may directly or indirectly affect ecosystem composition and structure (Noss 1990), such as the variety of species or the pattern of different habitat types. Reciprocally, alteration in ecosystem composition or structure may also influence its functions (Karr 1993). Thus, a degraded ecosystem shows impaired structures, composition and/ or functions (Figure 1).

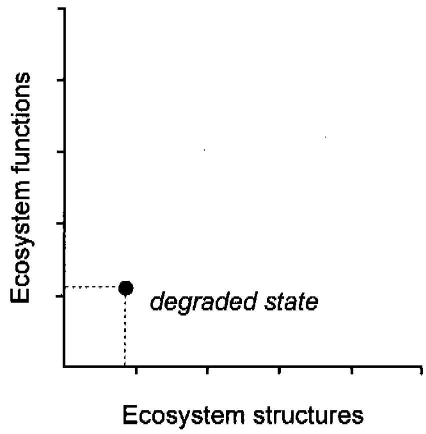

Figure 1: Degradation of an ecosystem expressed in terms of impaired structure and function (modified from Bradshaw 1997). For simplification, the third axis, i.e. ecosystem composition, is not displayed.

In its pristine state, an ecosystem is characterised by an intact, site-specific composition, structure and function (Figure 2; Jungwirth and others 2002). Such a system may serve as a reference to guide restoration activities (Leitbild or guiding image; Jungwirth and others 2002; Palmer and others 2005). The definition of reference conditions is the most crucial task in restoration (Angermeier 1997) and several approaches are proposed in the literature.

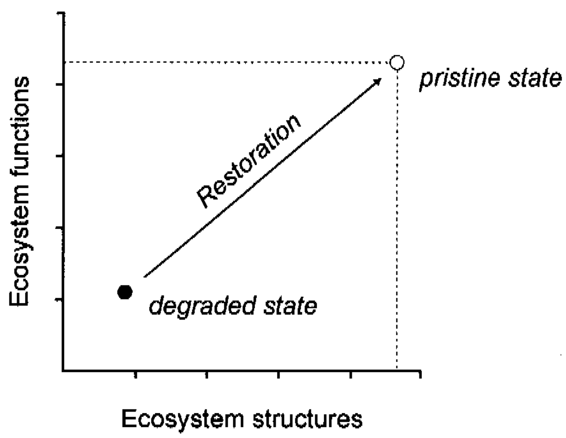

Figure 2: The process of restoration in which a reference state is reached reflecting pristine ecosystem structure and function (modified from Bradshaw 1997). For simplification, the third axis, i.e. ecosystem composition, is not displayed.

The use of contemporary reference systems is considerably hampered as there are only few natural or near-natural river reaches remaining in the industrialised countries worldwide (Benke 1990; Nilsson and others 2005; Ward and others 1999). Valuable alternatives are use of historic information (Hohensinner and others 2004; Jungwirth and others 2002; Kondolf and Larson 1995) or theoretical reconstruction based on conceptual or empirical models or classification systems (Palmer and others 2005). 'Restoration' (Figure 2) is generally defined as the process of bringing a system back to its pristine, intact state (Bradshaw 1997; Roni and others 2005) or historic trajectory (SER 2005).

In practice, however, a return to pristine conditions is often difficult or even impossible to achieve. Too many infrastructural or financial constraints exist, such as densely populated areas or communication networks (Stanford and others 1996). Thus, project objectives must be formulated by considering the present conditions with their opportunities and constraints (Nienhuis and Leuven 2001). Jungwirth and others (2002) refer to this process as the development of an 'operational Leitbild' (as opposed to the 'visionary Leitbild' characterising the undisturbed pristine conditions). 'Rehabilitation' describes the approach of improving ecosystem composition, structure and function to a near-natural, but not pristine state (Figure 3; Bradshaw 1997). As in restoration, this has to result in a system with integrity (Angermeier 1997).

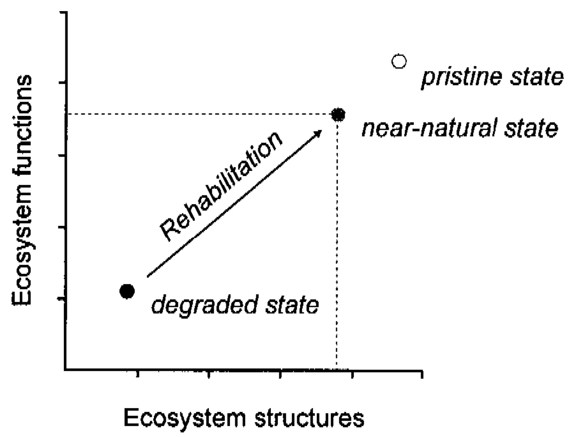

Figure 3: The process of rehabilitation in which ecosystem structures and functions are improved, but the pristine state is not totally achieved (modified from Bradshaw 1997). For simplification, the third axis, i.e. ecosystem composition, is not displayed.

„Restoration: an acid test for ecology"
Bradshaw (1987)

The number of realised river rehabilitation projects has increased continuously over the past 25 years, and river rehabilitation will remain an important subject on the international agenda (Bernhardt and others 2005). Rehabilitation projects offer a great potential for scientists from various disciplines to test and refine their understanding of riverine processes and functions (Bradshaw 1987). Vice-versa, these experiences should be transferred to the practice to further develop a technologically, ecologically and socially sustainable river management.

Two major rehabilitation programmes on the Swiss rivers Thur and Rhone were scientifically accompanied by a five-year transdisciplinary research project, the Rhone-Thur Project (Peter and others 2005), conducted by the Swiss Federal Institute of Aquatic Science and Technology (Eawag) and the Swiss Federal Institute for Forest, Snow and Landscape Research (WSL). This research project aimed to deliver scientific principles for both the two specific programmes and future rehabilitation projects in Switzerland.

The present thesis was part of the subproject "Fish Ecology" within the Rhone-Thur Project. Fish are highly suitable organisms for the detection of stream impairment and recovery due to their high mobility and longevity and the extensive knowledge available on their ecological role (Fausch and others 1990; Jungwirth and others 1995; Karr and others 1986). In this thesis selected spatio-temporal aspects of the rehabilitation process were investigated, focusing specifically on the evolution of the degradation and the assemblage recovery following rehabilitation. The thesis is divided into four individual studies which cover different stages of the rehabilitation process as described in Figures 1 to 3.

The evolution, distribution and relative influence of fish-ecological deficits were studied in the highly degraded Swiss Rhone system (**Chapter 2**). This was achieved by an analysis of the current and historical habitat and fish diversity at catchment level as suggested in the literature (Kondolf and Downs 1996). The precise description of present, degraded conditions (Figure 1) is an important, though frequently neglected element for planning future rehabilitation measures (Roni and others 2005).

Limited financial resources are often a central problem in nature conservation and rehabilitation. Setting management priorities is therefore a crucial task (Roni and others 2002). In **Chapter 3** we discuss a method to spatially explicitly identify priority areas for rehabilitation at a landscape scale. The method is primarily based on a map comparison between the pristine state (Figure 2) and the current degraded conditions (Figure 1).

Local river widening is a promising approach for the rehabilitation (Figure 3) of formerly braided rivers with still intact or little impaired bed load (Rohde and others 2004; Unfer and others 2004). The technical aim of this measure is to halt river bed erosion (Peter and others 2005). The seasonal habitat supply and fish assemblage structure in three local river widenings and adjacent canalised segments on the river Thur is documented in **Chapter 4**. Special emphasis was placed on the spatial setting of each segment within the river system and on the comparison with near-natural reference conditions (Figure 2).

River rehabilitation should initiate the recovery of abiotic conditions and biotic assemblages (Gore 1985; Kauffman and others 1997). However, many rehabilitation projects do not achieve the desired biological success (Roni and others 2005), implying that many essential aspects of the recovery process are still poorly understood. Recovery rates in riverine fish assemblages following habitat rehabilitation are discussed in a literature review in **Chapter 5**. Special emphasis is given to the spatio-temporal prerequisites for recovery and on fish-ecological indicators as well as end-points used in project evaluation (Figure 3).

In the synopsis (**Chapter 6**) the results from the individual studies are integrated and implications for rehabilitation practice are discussed.

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CHAPTER 2

Spatio-temporal analysis of fish and their habitat: a case study on a highly degraded Swiss river system prior to extensive rehabilitation

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Abstract

The failure of river rehabilitation projects is often reported in the literature. One possible reason for this failure is the insufficient consideration of factors degrading riverine ecosystems at large spatio-temporal scales. A precedent analysis of the evolution and significance of these factors at the watershed level is proposed as a prerequisite for a successful rehabilitation project. Based on a watershed-scale approach, we investigated the current and historical states of the fish assemblage and of relevant abiotic factors in the river Rhone, a seventh-order stream in Switzerland scheduled for large-scale rehabilitation. Recent field data gathered by electrofishing and habitat mapping were analysed by means of a mixed model approach and were qualitatively compared to historical information derived from topographic maps and documentary sources.

The length of the entire active channel has been reduced by 45 % (102 km) since 1850, representing a significant diminution in lateral connectivity. Our recent fish survey revealed a depleted species set, with only two of 19 historically documented species found. The density of brown trout was generally low, but positively correlated with the presence of cover. Thus, morphological improvements, e.g. through local river widening, offer extensive potential for the restoration of native fish assemblages, but will probably only be successful in combination with a more natural hydrological regime.

Keywords: Catchment; hydropeaking; rehabilitation; Brown trout; historical analysis; river Rhone.

Introduction

River management philosophy has changed fundamentally in recent years (Gleick, 2001). Today, it no longer focuses exclusively on total control and exploitation. On the contrary, environmental and socio-economic concerns are equally highlighted in many countries' legislation (e.g. the European Water Framework Directive), and both the conservation of pristine sites and rehabilitation of lost ecological structures and functions are seen as a priority.

However, this change in management philosophy is often very difficult to implement in practice. Today, 59 % of the world's large river systems are moderately to strongly affected by flow regulation, such as reservoir operation, irrigation or fragmentation by dams (Nilsson et al., 2005). In the United States only 2 % of rivers and streams are categorised as relatively natural and more than one third is considered as heavily impaired or polluted (Benke, 1990). The majority of lowland rivers in Europe is canalised (Brookes and Shields, 1996). In North America and Europe, 90 % of all floodplains have lost their ecological functioning due to cultivation (Tockner and Stanford, 2002).

Despite the completion of numerous rehabilitation projects, the positive effects of rehabilitation measures on riverine biota have rarely been documented, while the failure of several practical measures such as placement of log weirs (Frissell and Nawa, 1992), artificial riffles (Pretty et al., 2003), boulders (Lepori et al., 2005) or spawning gravel (Iversen et al., 1993) has been reported. The reasons for the biological failure could include failure to consider different spatio-temporal scales or confusion between them (Kondolf and Downs, 1996; Moerke and Lamberti, 2003). For financial or planning reasons, rehabilitation measures are often designed at a local level, neglecting processes and deficits operating at the watershed scale (Kondolf and Downs, 1996). Similarly, temporal aspects, such as pristine morphology, the natural flow regime (Poff et al., 1997; Richter et al., 1997) and the evolution of deficits over time, are often ignored in project planning (Jungwirth et al., 2002).

To improve rehabilitation practice, the factors that degrade a stream ecosystem need to be identified at an extensive spatio-temporal scale (Kauffmann et al., 1997; Lewis et al., 1996; Moerke and Lamberti, 2003), i.e. by carrying out an analysis of the current and historical state at the watershed level (Kondolf and Downs, 1996). Both of these aspects were addressed in the presented project. We studied the evolution, distribution and relative influence of physico-chemical deficits of relevance to fish along the river Rhone, a seventh-order stream in Switzerland. The river is scheduled to undergo large-scale rehabilitation as part of a flood protection project. Fish are highly suitable organisms for the detection of stream impairment and recovery due to their high mobility and longevity and the extensive knowledge available on their ecological role (Fausch et al., 1990; Jungwirth et al., 1995; Karr et al., 1986). Our watershed-scale approach is intended to achieve the following aims: 1) to demonstrate the importance of both spatially and temporally broader concepts that may be useful in rehabilitation practice and 2) to asses the potential to rehabilitate the fish assemblage of a hydrologically and morphologically impaired river system.

Materials and methods

Study sites

This study was conducted in the Swiss section of the river Rhone (Fig. 1) from its source at the Rhone Glacier (1763 m a.s.l.) to its mouth at Lake Geneva (374 m a.s.l.). Along this 167.5 km stretch, the river Rhone drains a catchment of 5,220 km^2 consisting mainly of forest and pastures (46 %), rocks (24 %) and glaciers (14 %), and agricultural land (16 %) (Loizeau and Dominik, 2000; Spreafico et al., 1992).

The river Rhone has been considerably altered over the past two centuries, morphologically due to two corrections of the river corridor carried out in the late 19^{th} and early 20^{th} centuries (Département Fédéral de l'Intérieur, 1964) and also as a result of hydrological alterations arising from the exploitation of hydropower in the catchment (Loizeau and Dominik, 2000). Today, the Rhone is mainly channelized and river reaches with a near natural flow regime are rare. The mean natural annual discharge has been reduced by more than 20 % along 22 % (36 km) of the entire river course between the source and Lake Geneva. Hydropeaking prevails over a distance of 109 km (65 % of the entire river course; data from Spreafico et al., 1992).

In most reaches affected by hydropeaking, winter flows are significantly increased (Meile et al., 2005; Spreafico et al., 1992) due to the predominant release of water during the winter months.

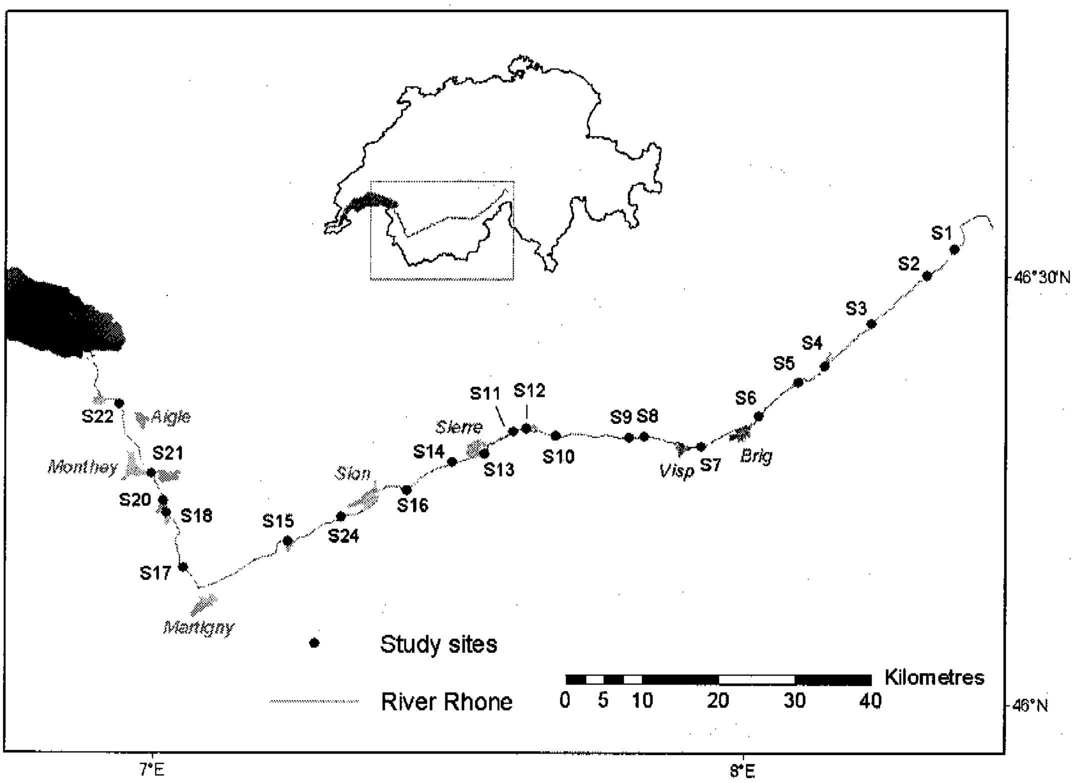

Figure 1: Location of the river Rhone upstream from Lake Geneva in southern Switzerland (© 2003 swisstopo). Points indicate the study sites.

It is planned to renovate and improve existing flood protection measures along a significant part of the river Rhone (69.9 km) over the next 25 years. A simultaneous improvement of ecological and socio-economic concerns is also planned. The aim is to re-create a diverse, more naturally functioning river that will also provide a popular recreation area for the local population and tourists.

Under current conditions, the river Rhone is categorized as a trout-grayling zone (Huet, 1959). Considerable stocking of brown trout with both young-of-the-year and adult trout exceeding the legal limit size of 240 mm is carried out along the entire river course. The annual stocking of the latter averages around 5 tons (R. Collaud, personal communication).

Study sites were selected by stratified random sampling (Krebs, 1989). This method allows for possible heterogeneity within the sampled population due to variation in environmental attributes (Jeffers, 1998). We divided the river Rhone into 18 segments or strata on the basis of topographical (slope), hydrological (influencing of the discharge volume by hydropower generation) and morphological (bank type) data. Sites accessible with field equipment were determined randomly within each stratum. Because of the difficult topography, 7 shorter strata with a total length of 10.7 km (6.4 %) could not be sampled. Altogether, 22 sites located in 11 strata were included in the investigation (Fig. 1).

Fish parameters

Each randomly determined site was electrofished once using a semiquantitative approach (one pass, without stop nets). The survey took place in February and March 2003 under winter, low flow conditions. A stationary electroshocker was used in most cases (EFKO, 8 kW, 150-300/ 300-600 V). Stretches that were difficult to access were fished using backpack electrofishing equipment (EFKO, 1.5 kW, 150-300/ 300-600 V).

Fishing was conducted on strips of the riverbed of at least 100 m length. In the lower part of the river, sampling was restricted to the banks; a strip in the mid-channel was also included at sites with minor discharge. Bank and mid-channel strips were 3.2 m wide on average. Narrow reaches in the headwaters were fished over the entire width. A total of 36 strips were fished over the 22 sites, and the fished area was determined by multiplying strip length and width.

The fish were handled in accordance with a standardized procedure (controlled conditioning, anaesthesia with clove oil (Hänseler AG, Herisau, Switzerland; 0.5 mL diluted in 9.5 mL alcohol added to 20 L water)). Body length (±1 mm) and the presence and type of any anomalies were determined. All of the fish were released along the fished stretch after recovery.

The following biological parameters were determined from the catches: density of individuals in total and per species [individuals $\cdot$ m^{-2}], median length of brown trout [cm] and percentage of brown trout with anomalies and injuries.

Environmental variables

The strips were divided into 10 sectors of equal length for measuring environmental variables. Substratum composition was estimated in four randomly selected sectors (Bain et al., 1985) and assigned to one of nine classes using a modified Wentworth scale (Cummins, 1962). Aquatic habitats (Hawkins et al., 1993) were mapped and their percentages visually estimated in each sector. Based on this mapping, habitat diversity was determined using Shannon's index of diversity and evenness (Arscott et al., 2001; Matthews, 1998). Presence and type of suitable fish cover were determined visually, i.e. the area providing shelter from predators and high current velocities was identified. Both overhead cover and slow water areas behind submerged objects were considered in accordance with Peter (1992). Shoreline composition regarding particle type and size was recorded.

Water samples were taken at each site directly after electrofishing (one sample per site) and immediately deep frozen in dry ice. In the lab, the samples were thawed and analysed for total phosphorous (Tot-P), NO_3-N, NO_2-N, NH_4-N, pH, alkalinity, dissolved organic carbon (DOC), total organic carbon (TOC) and suspended solids according to standard methods.

The impact of hydropower installations on each study site was expressed using selected hydrological parameters based on the hydrological atlas of Switzerland (Spreafico et al., 1992): the degree of water diversion was described by the percentage of the mean natural annual discharge remaining. The impact of hydropeaking on the seasonal discharge regime was quantified as a percentage increase in the natural winter flow.

Statistical analysis

Abiotic factors were analysed by means of principal components analysis (PCA) to identify environmental variables that varied most between the sites. This procedure involves the creation of uncorrelated groups of intercorrelated variables, which are referred to as principal components. In order to determine the relationship between environmental variables and biotic parameters, all principal components with an eigenvalue > 1 were compared with the fish parameters using a linear mixed-effects model procedure (SPSS Inc., 2005). This method allows the inclusion of both random and fixed factors as well as the consideration of experimental units that are nested in a hierarchy (i.e. several strips per site).

The analysis was conducted using the SPSS 11.0.1 for Windows software package. Prior to the analysis, all data were transformed using standard transformations (arcsin, log, square root).

Historical analysis

For quantification of the former morphological state of the river Rhone, historical maps from 4 different periods (1850, 1900, 1950 and 2003) were georeferenced and digitized by using ArcMap™ 8.3 (ESRI). With the exception of a small region on Lake Geneva in 1900 (1:25,000) and all of the 1850 maps (1:100,000), most of these maps were at a scale of 1:50,000. Due to the limited availability of suitable maps, the historical analysis was restricted to the region between Brig and Lake Geneva. Four measures quantifying important riverine characteristics were calculated on the basis of the digitizing: 1) The total length of the main channel [km], and 2) the total length of the entire active channel [km] characterize the river type and structure; 3) the wetted width of the active channel [m] gives information about flow patterns; and 4) to identify the degree of lateral connectivity, the shoreline length was measured in kilometres of shoreline per river kilometre (van der Nat et al., 2002).
Due to limitations in data availability, it was not possible to carry out a more in-depth analysis of historical fish data. An estimate of the potential fish fauna was carried out from available historical sources (Fatio, 1882; 1890; Gattlen, 1955).

Results

Historical analysis

The main channel length of the river Rhone between Brig and Lake Geneva has been reduced by 4.6 km or 3.7 % over the past 150 years (Table 1). This difference increases substantially (-102.2 km, -44.7 %) when the length of the entire active channel (cumulative length of all branches) is considered. The most extensive decrease took place between 1850 and 1900. The length of the main channel and of the total active channel has been more or less stable since then.

Table 1: Temporal evolution of four riverine features relevant to fish ecology and the fish-biological zonation for a section of the river Rhone (between Brig and Lake Geneva).

	Year			
	1850	1900	1950	2003
Length of the main channel (km)	123.60	119.34	119.25	118.98
Total length of the active channel (km)	228.80	133.37	132.44	126.59
Shoreline length (km)	414.41	264.40	257.09	250.60
(km/km main channel length)	3.36	2.22	2.16	2.11
Mean wetted width (median in m)	93	53	69	53
Fish zone	Trout – Grayling – Zone			

The shoreline length of the river Rhone between Brig and Lake Geneva decreased by 39.5 % (163.8 km) between 1850 and 2003, and the shoreline length per river kilometre was reduced from 3.4 to 2.1. The most significant change in shoreline length took place between 1850 and 1900, with a decrease of 150.0 km (36.2 %), indicating a decline from 3.4 km $\cdot$ km^{-1} to 2.2 km $\cdot$ km^{-1}.

The median wetted width has been reduced by 40 m or 43 % since 1850, with the most important reduction also occurring between 1850 and 1900. This decline was interrupted by a temporary increase in the median wetted width in 1950.

Based on the morphometric conditions retrieved from the historical maps, the river Rhone was categorized as a trout–grayling zone (Table 1). Eight fish species were mentioned in the oldest available documentary source dating from 1544 (Table 1; see Gattlen, 1955). In the late 19th century, 19 fish species were documented in the river Rhone and its tributaries (Fatio, 1882; 1890).

Table 2: Fish species in the river Rhone reported in historical sources (columns 1-2) and in selected recent literature (columns 3-5).

	1	2	3	4		5
				Tributaries	Canals	
Salmo trutta fario	x [a)]	x	x	x	x	x
Salmo trutta lacustris		x				
Coregonus spp.		x				
Oncorhynchus mykiss			x		x	
Salvelinus fontinalis			x			
Thymallus thymallus	x	x	x			
Cottus gobio	x	x	x	x	x	x
Phoxinus phoxinus		x	x	x	x	x
Alburnus alburnus		x	x			
Gasterosteus aculeatus			x		x	
Esox lucius	x	x			x	
Leuciscus cephalus	x	x			x	
Perca fluviatilis		x				x
Gobio gobio	x [b)]	x				x
Carassius auratus						x
Cyprinus carpio	x	x				
Barbatula barbatula	x [b)]	x				
Tinca tinca	x	x				
Rutilus rutilus		x				
Scardinius erythrophthalmus		x				
Lota lota		x				
Alburnoides bipunctatus		x				
Anguilla anguilla		x				
Total:	8	19	8	3	7	5

1 Sebastian Münster's Kosmographie from 1544 (see Gattlen, 1955):
a) inclusion of subspecies unclear. b) common name unclear, gudgeon or stone loach.

2 Fatio (1882) and Fatio (1890)

3 Etat du Valais (1999)

4 Küttel (2001): Electrofishing survey in selected tributaries and canals of the Rhone valley.

5 Peter (2004)

Fish parameters

Our electrofishing surveys revealed a low diversity of fish species in the river Rhone. Only two species, brown trout (*Salmo trutta fario*) and bullhead (*Cottus gobio*), were found in the 36 strips sampled. The relative abundances showed a high dominance of brown trout, amounting to 99.6 % (714 individuals) of the total catch (717 individuals). Generally, very low densities were found, ranging from 0 in 4 strips to 43 fish$\cdot$100 m^{-2}, with a mean of 5 fish$\cdot$100 m^{-2}.

Most brown trout were of medium body size. The average length per strip (median) ranged between 87 mm and 261 mm with a mean of 157 mm. Larger and young individuals were largely missing. A considerable number of brown trout (28 %) showed anomalies such as deformed fins and shortened opercula, as can often be found in hatchery-reared animals.

Environmental variables

Table 3 summarizes the distribution of the untransformed original environmental variables. The values of all the chemical parameters were within the tolerable range for brown trout (Alabaster and Lloyd, 1980).

Table 3:: General statistics for environmental variables used for the Principal Component Analysis (PCA). The values are calculated on the basis of the individual fishing strips (N = 36).

Environmental variable	Unit	Median	Minimum	Maximum	Interquartile Range
Substratum < 8mm	%	0.00	0.00	100.00	31.25
Substratum 8-64mm	%	5.00	0.00	100.00	50.00
Substratum 64-256mm	%	50.00	0.00	100.00	41.67
Substratum > 256mm	%	0.00	0.00	50.00	25.00
Cover availability	%	2.40	0.00	36.57	6.16
Riffles	%	22.80	0.00	91.70	46.63
Glides	%	11.00	0.00	87.50	29.08
Runs	%	12.74	0.00	100.00	49.63
Edgewaters shallow	%	7.50	0.00	41.50	17.62
Pools and deep edgewaters	%	11.35	0.00	100.00	23.82
Number of habitat types	count	5.00	1.00	11.00	3.00
Evenness	nondimensional	0.75	0.00	0.99	0.20
Shanon index of diversity	nondimensional	1.71	0.00	2.89	0.68
Shoreline fine	%	0.00	0.00	77.00	0.00
Shoreline mixed	%	0.00	0.00	54.50	0.00
Shoreline organic	%	0.00	0.00	52.75	0.00
Shoreline coarse or rock	%	100.00	0.00	100.00	70.25
Residual flow	nondimensional	9.00	1.00	10.00	5.00
Increased winter flow	nondimensional	1.00	0.00	3.00	2.00
[Tot-P]	$\mu g \cdot L^{-1}$	20.60	5.10	85.60	21.65
[NO_3]	$mg \cdot L^{-1}$	0.68	0.34	1.10	0.47
[NO_2]	$\mu g \cdot L^{-1}$	8.55	1.00	25.80	9.55
[NH_4]	$\mu g \cdot L^{-1}$	109.35	5.00	519.00	160.70
pH	nondimensional	7.34	7.06	7.70	0.27
Alkalinity	$mmol \cdot L^{-1}$	1.46	0.68	2.65	0.87
Suspended solids	$mg \cdot L^{-1}$	15.71	3.67	65.30	8.03
DOC	$mg \cdot L^{-1}$	0.67	0.43	1.12	0.24
TOC	$mg \cdot L^{-1}$	0.79	0.54	2.11	0.69

In the principal components analysis (PCA), the 28 original variables were reduced to 9 principal components, accounting for 85.6 % of the total variation (Table 4). Every original variable displayed high loading (> 0.5) in a single principal component only, thereby enabling clear interpretation. The variables "(abundance of) riffles" and "(concentration of) suspended solids" did not show any loadings > 0.5 (Table 4).

Table 4: Loadings of environmental variables on principal components (PC) from PCA. Principal components with eigenvalues > 1 were extracted. Loadings > 0.5 are shown in bold print.

Environmental variable	PC1	PC2	PC3	PC4	PC5	PC6	PC7	PC8	PC9
$[NO_3]$	**-0.86**	-0.09	0.31	-0.03	-0.11	-0.09	-0.05	0.13	0.14
Residual flow	**0.85**	-0.01	0.29	-0.08	0.11	0.03	-0.21	-0.02	0.01
Runs	**0.82**	-0.30	0.02	0.10	0.03	-0.04	-0.24	0.04	-0.15
[Tot-P]	**0.77**	0.01	0.38	-0.15	0.27	-0.03	0.29	0.19	0.02
Increased winter flow	**0.72**	-0.10	0.47	-0.19	0.01	0.16	-0.24	0.00	0.09
pH	**-0.51**	0.21	0.05	-0.23	0.16	-0.33	0.15	0.35	-0.23
Alkalinity	**-0.73**	-0.29	-0.05	-0.24	-0.23	-0.03	-0.10	0.19	0.15
Shanon index of diversity	0.01	**0.94**	-0.03	0.02	0.14	0.04	0.03	-0.03	0.22
Evenness	-0.10	**0.80**	0.35	-0.14	-0.07	-0.07	-0.26	0.01	0.09
Number of habitat types	0.08	**0.75**	-0.41	0.23	0.22	0.03	0.23	-0.08	0.06
$[NO_2]$	0.01	0.03	**0.95**	-0.08	0.09	-0.04	0.05	0.02	0.02
$[NH_4]$	0.42	-0.02	**0.84**	-0.12	0.07	-0.09	0.11	0.14	-0.01
DOC	0.02	0.00	-0.01	**0.93**	-0.08	0.09	0.09	0.09	0.02
TOC	0.06	0.02	-0.21	**0.86**	-0.01	0.04	0.12	0.06	0.03
Substratum < 8mm	0.15	0.14	0.02	0.13	**0.89**	-0.04	-0.25	-0.12	0.09
Shoreline fine	0.21	0.08	0.13	-0.32	**0.87**	-0.09	0.07	0.01	-0.07
Pools and deep edgewaters	0.00	-0.33	0.09	-0.15	0.24	**0.78**	0.05	-0.22	0.10
Substratum > 256mm	-0.03	0.19	-0.27	0.21	-0.21	**0.71**	0.00	0.09	0.00
Cover availability	0.21	0.25	0.40	-0.02	-0.06	**0.66**	0.02	0.03	-0.39
Glides	-0.41	0.26	0.22	-0.19	0.09	**-0.53**	-0.05	-0.24	0.11
Substratum 8-64mm	-0.20	-0.07	0.09	0.08	-0.27	-0.06	**0.88**	-0.12	0.04
Substratum 64-256mm	0.03	-0.15	-0.06	-0.22	-0.50	-0.26	**-0.67**	0.16	-0.04
Shoreline mixed	0.02	0.04	-0.09	-0.35	0.13	0.01	0.34	**-0.74**	-0.08
Shoreline organic	0.17	0.09	-0.43	0.44	0.01	-0.19	-0.05	**-0.55**	-0.13
Edgewaters shallow	-0.22	0.43	0.06	-0.03	0.12	-0.14	0.09	0.04	**0.76**
Shoreline coarse or rock	-0.03	0.30	0.04	0.25	-0.32	0.49	-0.05	0.25	**0.54**
Riffles	-0.28	0.47	-0.38	0.20	-0.21	-0.02	0.35	0.33	-0.25
Suspended solids	0.35	-0.25	0.07	-0.49	0.46	-0.15	0.30	0.15	0.04
Eigenvalue	4.80	3.22	3.17	2.84	2.64	2.42	2.07	1.48	1.33
% total variation	17.14	11.49	11.30	10.13	9.42	8.64	7.38	5.29	4.76

The extracted components can be labelled in accordance with the grouping of the original variables. Component 6, for instance, can be referred to as presence of cover, with variables like availability of cover, larger substratum and pools being the most important (see the high positive loadings in Table 4). In addition, the percentage of glides is negatively correlated with component 6.

Component 4 summarizes the organic content of the water sample while component 7 includes the availability of middle-sized substratum (positive loading for particles between 8 and 64 mm, negative loading for larger particles).

Comparison between abiotic and biotic variables

The linear mixed-effects model procedure revealed a significant positive relationship between component 6 (cover availability) and two biotic variables (Table 5). Thus, large numbers of brown trout were found in stretches with a considerable amount of cover, substratum larger than 25 cm in diameter, a high percentage of pools and a small percentage of glides.

It was not possible to establish any relationship between the environmental variables and the percentage of anomalies and injuries found on the brown trout.

Table 5: Relationship between environmental and fish-biology variables (results from the linear mixed-effects model procedure).

Fish-biology variable	Principal component								
	1	2	3	4	5	6	7	8	9
Mean length of brown trout (median)	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Percentage of brown trout with anomalies	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Percentage of brown trout with injuries	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Number of fish $\cdot$ m^{-2}	n.s.	n.s.	n.s.	n.s.	n.s.	$p = 0.000$ $F = 24.34$	n.s.	n.s.	n.s.
Number of brown trout $\cdot$ m^{-2}	n.s.	n.s.	n.s.	n.s.	n.s.	$p = 0.001$ $F = 24.24$	n.s.	n.s.	n.s.

Discussion

Morphological and hydrological deficits

In the mid-19th century, the Rhone system was still in a near-natural state with isolated human modifications of limited spatial scope. Despite this low degree of structural impairment in 1850, morphological characteristics such as the shoreline length clearly differ from contemporary values obtained in the Tagliamento, NE Italy, i.e. the only large morphologically intact Alpine river remaining in Europe (Ward et al., 1999). There, the shoreline length measured in the field averaged 14.4 km $\cdot$ km^{-1} (van der Nat et al., 2002). Our values are substantially lower. This discrepancy is most probably due to differences in data collection and spatial resolution (high resolution dGPS vs. on-screen digitization of historical maps).

To our knowledge, shoreline length has never previously been measured from historical maps. A critical problem in the comparison of various temporal states is that potential differences may indicate variations in discharge rather than changes in river morphology. Unfortunately, no information is available on the discharge conditions at the time of the cartographers' work. This must be considered in the interpretation of the data. However, from the inundation pattern in the last remaining braided section in the Rhone (see below), it can be assumed that none of the four states displays extreme discharge conditions, i.e. neither extremely low nor extremely high flows and that conditions are more or less comparable.

The geomorphological characteristics of the river Rhone have changed fundamentally over the past 150 years. This development is reflected in a slight decrease in main channel length and a dramatic reduction in shoreline length, total active channel length and, partly, median wetted width. The most significant changes between 1850 and 1900 coincide with the systematic straightening of the river Rhone in the 1860s and 70s (Département Fédéral de l'Intérieur, 1964).

Lateral connectivity was most strongly affected by channelization. Characteristic aquatic habitats, such as backwaters, side arms, oxbow lakes and marshes, completely disappeared or are now permanently uncoupled from the main channel, even during severe floods.

With the loss of the natural riparian zone, a fundamental component of the river was reduced and impaired, affecting a variety of physical and ecological functions such as habitat structure, organic matter supply, etc. (Naiman and Décamps, 1997; Schiemer and Zalewski, 1992). The negative effects of reduced lateral connectivity on fish stocks are well known (Welcomme, 1979).

Today, only one 6 km reach of the river Rhone between Brig and Lake Geneva still shows a nearly natural braided structure (Pfynwald, upstream from Sierre, Fig. 1). However, despite this physical connection with the floodplain, the natural flow dynamics cannot develop to the maximum extent as an upstream water diversion causes residual flow throughout the year. Longer near-natural braided river reaches with extensive floodplains can still be found below Lake Geneva, in the French section of the river Rhone (Copp, 1989).

Instream structure has also been drastically reduced: prior to the first correction, a high variability in stream width – and with it probably in depth - resulted in a diverse physical habitat, an effect that was probably amplified by the large woody debris deriving from the vast riparian forests indicated on the topographic maps. The systematic straightening of the river Rhone resulted in the development of a barely structured, uniform bed profile. Furthermore, due to the changes in land use and the removal of snags at the weirs, the structuring effect of large woody debris is now almost completely lacking. As a result, habitats with high current velocities and uniform depths prevail today. Habitats with slow flowing or even standing-water like pools are strongly underrepresented. Such a shift from lentic to lotic conditions represents a typical development in channelized rivers (Welcomme, 1979).

Changes in channel morphology can also be assumed to alter water temperature (Hawkins et al., 1997). In a naturally structured system, temperature variation in the cross-section can exceed that occurring in the main channel along the entire river course (Arscott et al., 2001). Standing water bodies of various connectivity with interstitial waters (phreatic or hyporheic) are responsible for much of this lateral temperature heterogeneity (Arscott et al., 2001). In a channelized river like the river Rhone, however, almost no lateral heterogeneity in water temperature exists.

Temperature is known to be a key environmental factor structuring fish communities (Welcomme, 1979). In particular, the requirements of meso-eurythermal species like the European chub (*Leuciscus cephalus*) or the gudgeon (*Gobio gobio*) are not fulfilled under the present homogeneous temperature conditions prevailing in the river Rhone.

Deficits in the fish fauna

Assemblage level

Historical cartographical and documentary sources show a diverse habitat occupied by a relatively rich fish assemblage. Against this background, our recent survey revealed an extremely low species richness in the river Rhone, with only two species present. Moreover, the fish fauna is almost completely composed of brown trout, with a few bullheads accounting for < 0.5 % of the total catch. Previous local studies, partly conducted at confluences of tributaries, revealed a higher diversity, i.e. 6 and 8 species, respectively, but most species were represented by single individuals (Etat du Valais, 1999; Peter and Weber, 2004). Similar results were reported for the tributaries (3 species, low densities), whereas a slightly larger species pool of 6 species and higher fish densities were found in the channels of the Rhone Plain (Küttel, 2001). A richer species set is reported from the French section of the river Rhone, below Lake Geneva (see e.g. Copp, 1989; Persat et al., 1994), where near-natural floodplains exist that are still large and have a rich habitat supply.

The low species diversity and unnatural assemblage composition in the main river constitutes a serious biotic deficit. In our search for possible reasons, we confined ourselves to a qualitative interpretation. Because of the dominance of the brown trout, the statistical analysis had to be restricted to the population level of this species (see the following section). As described in the preceding chapter, the diversity of aquatic habitats in the river Rhone has suffered massive impairment. This morphological degradation is further compounded by hydrological alterations related to hydropower generation. Five dams on the main river and a large number of weirs in the tributaries interrupt longitudinal connectivity and inhibit the passage of migratory species such as the lake resident trout (*Salmo trutta lacustris*).

The effects on the flow regime are also considerable. Extensive variability in width and depth can often be observed in stretches under residual flow. Because of the reduced discharge volume, however, the water column is very small, preventing larger fish from colonizing these sites. Furthermore, reservoir management has completely changed the daily and seasonal discharge patterns. The natural winter flow has increased substantially, and daily flow fluctuations lead to a general increase in current velocities and a lateral displacement or even disappearance of rare lentic zones (Bain et al., 1988). Species or age classes limited to this habitat are forced to relocate, and some individuals, especially smaller fish, may suffer from stranding (Saltveit et al., 2001).

Population level: brown trout

The observed densities of brown trout are very low for a river of the trout-grayling zone. Moreover, the age structure represented in the catches differs clearly from that of a natural population. Young trout (0+, 1+) are highly underrepresented in all sampled stretches. This observation is reinforced by former surveys that specifically documented the occurrence of brown trout fry shortly after emergence (Peter and Weber, 2004). Natural 0+ brown trout could only be found at isolated sites. There are several reasons for these small cohort sizes. Firstly, adult, reproductive animals are only found in very low densities. Secondly, our habitat surveys indicate that both spawning and rearing conditions are unsuitable. As mentioned above, the habitat preferred by young trout, i.e. shallow riffles with cobble substrata (Heggenes, 1988), is generally sparsely available in the river Rhone and, because of the flow fluctuations induced by hydropeaking, often of reduced persistence. The effects of these unstable habitat conditions, in particular on fry and juvenile trout (e.g. stranding, increased drift), are widely reported in the literature (Freeman et al., 2001; Hunter, 1992; Liebig et al., 1999; Saltveit et al., 2001).

The reproductive contribution of individuals inhabiting the tributaries is also of minor importance. Tributaries often lack a connectivity to the main river, and the impaired quality of spawning grounds and low densities of young fish (0+) have been documented (Küttel, 2001).

Although the brown trout densities observed in our study are low, variations could be observed between the different strips and sites. Some general relationships were observed in relation to the environmental variables. In general, more brown trout were caught where cover was available. This fact was particularly noticeable at sites where several strips with different cover availabilities were fished.

Cover in the river Rhone is mostly provided by artificial structures, in particular embankments. The dominant embankment type is riprap, followed by groins dating mainly from the first correction. Natural cover structures like woody debris, macrophytes or instream structures are largely missing. Cover is strictly reduced to the shoreline zone and is almost absent in the middle of the river. Accordingly, stretches fixed with riprap performed best in respect of cover availability. Both the highest brown trout densities and the highest biomasses were found in these stretches (Fette and Weber, 2007). Contrary to this, brown trout were absent from or only represented by single individuals along sparsely structured sandy shores secured by groins and in the monotonous median strips.

The large number of brown trout showing anomalies prompts the assumption that a significant proportion of the stock is hatchery-reared fish. Negative consequences for reproduction, the state of health and genetic composition of wild populations may be expected as a result (White et al., 1995).

Ecological potential and rehabilitation measures

Flood protection and river rehabilitation are no longer regarded as controversial in many countries (Nienhuis and Leuven, 2001). On the contrary, synergies have been identified that give rise to benefits for both. This is also true of the river Rhone. The third correction will change the face of the river considerably over the next 25 years. What priorities can be identified based on this study?

The most obvious structural deficits are the low quality of shoreline zones and the lack of instream structures such as pool-riffle-glide patterns. Under present conditions, linear banks fixed with riprap perform best in the promotion of brown trout density. They offer cover, but are not as diverse as well-structured near-natural shorelines (Schiemer and Spindler, 1989; Schmetterling et al., 2001). Many ecological functions would benefit from an improvement of the shoreline zone, and socio-economic services would increase (Hostmann et al., 2005; Naiman and Décamps, 1997; Tockner et al., 2008).

One measure that would enhance both the shoreline and the low instream habitat diversity and quality in formerly braided rivers is local river widening (Rohde et al., 2005). This would involve a significant widening of the river bed (Habersack et al., 2000), possibly by a factor > 2 of the existing channel width, in order to re-create a braided river morphology. Apart from river-engineering advantages – stabilization of the bottom by decreasing the transport capacity of the widened channel (Hunzinger, 1998) – local river widening offers many ecological improvements through morphodynamical processes such as braiding and gravel erosion and deposition, thus leading to increased structural and hydrological heterogeneity. Up to now, local river widening has mainly been carried out in Switzerland, Germany and Austria (Habersack and Nachtnebel, 1995; Rohde et al., 2005; Völkl et al., 2002).

Few studies are available on the ecological success of local river widening, however their findings are generally positive. Habersack (2000) describes the enhancement of aquatic habitat conditions and the increased stock of juvenile grayling in local widenings of the river Drau, Austria. A comparison of the riparian vegetation in 5 river widenings in Switzerland revealed an increased degree of naturalness at both habitat and species level in most of the cases studied (Rohde et al., 2004; Rohde et al., 2005).

Local river widening under impaired hydrological conditions as found in the river Rhone is particularly delicate and challenging (Unfer et al., 2004). In most cases, purely morphological measures will probably not compensate for hydrological deficits. Hydrological actions such as the mitigation of hydropeaking using technical measures are also necessary, e.g. slower ramping rates of the turbines (Halleraker et al., 2003) or storage of the turbinated water in retention basins (Moog, 1993). This combination of structural and hydrological measures alone can bring the anticipated success in degraded rivers with appropriate water quality and existing bedload dynamics (Fette and Weber, 2007).

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CHAPTER 3

Evaluation of the nature conservation value of the landscape over time: implications for rehabilitation projects

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Abstract

In general, nature is undergoing gradual ecologically-negative change and landscape restoration is set to become a very important field in the future. In this paper, we document human-driven land-cover changes in an originally river-dominated region of Switzerland, the Rhone plain, which developed from a near-natural reference state in the early 20th century to a highly anthropogenic state in the early 21st. We present a method that uses land-cover data and expert knowledge to enable a spatially explicit assessment of changes in the nature conservation value of the landscape over time. Our results suggest that human activity has had a negative effect on the ecological state of over 97% of the plain since 1900. The hot spots which experienced the most negative ecological changes are related to wetland destruction. Our study proposes a methodological GIS-based approach for determining and locating the rehabilitation potential of the landscape. The method enables the rapid, ecologically relevant and spatially complete evaluation of a large and heterogeneous landscape and could constitute an important tool for communication, landscape decision-making and biological conservation management in landscape planning.

Keywords: Land cover changes, landscape history, reference state, fragmentation, GIS, Switzerland.

Introduction

Nature is undergoing changes which are mostly negative, anthropogenic in origin, ominously large in scale and accelerating (Balmford and Bond, 2005). In addition to the risk of species extinction arising from climate change (Thomas et al., 2004), anthropogenic habitat fragmentation and loss are recognized as major contributors to the general decline and future threat to biodiversity (Pimm and Raven, 2000; Tilman et al., 1994). If the world's biological richness is to be conserved, natural and healthy ecosystems must be protected and recreated (Dobson et al., 1997). Given the limited financial, temporal and energy resources available, it is essential that priorities be set for conservation and restoration activities.

The conservation value of the landscape

Nature conservation can be defined as the preservation and protection of the natural richness of a landscape (i.e. soil, geomorphology, flora and fauna). The process of assessing the significance of an area for nature conservation is called ecological evaluation (Spellemberg, 1992). Thus, the conservation value of a landscape represents its capacity to ensure the persistence of natural richness over the time. According to these definitions, the performance of an ecological evaluation basically involves classifying the area under analysis into units of varying significance in terms of nature conservation (Geneletti, 2002), i.e. different conservation values. This requires a general evaluation concept which specifies both the conservation objectives and the criteria that express their fulfilment. Conservation objectives range from species-centred approaches focusing on the conservation of one or several endangered taxa to more process-oriented procedures aimed at the maintenance of functioning self-sustaining ecosystems (Geneletti, 2002; Margules and Usher, 1981; Noss et al., 1997; Spellemberg, 1992). In recent decades, new evaluation criteria (e.g. connectivity, patch shape) have been related to the field of landscape ecology as it addresses the relationship between spatial patterns (landscape composition and configuration) and ecological processes (Forman and Godron, 1986), therefore forming the base for the survival of species (Bridgewater, 1993; Burke, 2000; Hansson and Angelstam, 1991).

Conservation value may vary spatially in a fragmented and heterogeneous landscape. Natural or near-natural ecosystems such as wetlands, forests or alluvial river systems are more significant in terms of conservation than intensively exploited or impaired components of the landscape matrix, such as agricultural lands, urban areas and road networks. However, the conservation value of the landscape also varies temporally, in particular as a result of human activities (Forman and Godron, 1986). The rationalization and intensification of agricultural practices in Switzerland during the second half of the last century had an extremely negative impact on the capacity of the agricultural area to ensure the viability of a wide variety of species (Broggi and Schlegel, 1990, 1998). Former natural structures such as trees, hedges and shrubs have either disappeared or have become isolated. Thus, temporal landscape dynamics are among the most important characteristics that describe a landscape (Forman and Godron, 1986) and may have important implications for restoration projects (Egan and Howell, 2005a).

Reference state and rehabilitation

The quantification of temporal landscape dynamics requires the definition of a reference state that describes the landscape in its pristine and generally less disturbed state. Several techniques can be used to establish the reference conditions of an ecosystem or a landscape. These include the study of historic records (e.g. written and oral histories, photographs, maps) and the analyses of proxy records derived from biological sources such as pollen, spores or macrofossils (White and Walker, 1997). A reference state can support decision-making in landscape planning and biological conservation management and can also act as model or target for planning restoration projects, in particular when current conditions are seriously degraded and differ significantly with respect to the original state (Axelsson and Ostlund, 2001; Christensen, 1997; Gordon et al., 1997; Hohensinner et al., 2004; Jungwirth et al., 2002; Luyet, 2005; Nordlind and Ostlund, 2003; SER, 2004; White and Walker, 1997).

Nevertheless, historical analyses are generally carried out in places that have already been identified by previous investigations as suitable for restoration. Today, such reference conditions are used to drive restoration projects and not to locate them (Egan and Howell, 2005b).

The development of standardized methods which support decisions regarding the identification of areas most suitable for restoration at landscape and regional level is a recent and increasingly interesting research field (Hobbs and Norton, 1996). Due to the increasing complexity of the ecological systems being studied, it is even more difficult to decide at a broader level what should be restored, where and how (Hobbs and Harris, 2001).

In this study we used reference conditions as a model for the identification of locations in a given degraded and heterogeneous landscape where rehabilitation projects should take priority. Because it is recognized that ecological restoration will not necessarily translate into the re-establishment of the exact former state of an ecosystem or a landscape, we prefer to use the term rehabilitation (SER, 2004). Rehabilitation shares with restoration a fundamental focus on historical or pre-existing landscapes as models or references, however the goal is not the exact reconstruction of the original state.

Purpose

In this paper, we study the reference conditions and transformation over time of an originally river-dominated region of Switzerland, the Rhone plain. The objectives of the study are: (i) to develop a feasible method for the assessment of the conservation value of the landscape in the Rhone plain for the years 1900 and 2003 on the basis of land-cover maps and expert knowledge; (ii) to locate hot spots where landscape transformation has mostly reduced the conservation value of the plain since 1900; and (iii) to identify potential priority areas for landscape rehabilitation. It is our view that our approach could provide a useful starting point for the definition of conservation goals and for ecological rehabilitation projects in which the target state is based on an historical reference.

Material and Methods

Study area and land cover changes

The area studied is the section of the Rhone valley in Switzerland between Brig (678 m a.s.l.) and the mouth of the river Rhone into Lake Geneva (374 m a.s.l.) (Figure 1). The total area involved is 240 km^2. The length of the river, which is now almost completely canalized, in that area is approximately 120 km. The river Rhone drains a catchment area of 5,520 km^2 close to the mouth into Lake Geneva, and has an average annual discharge of 187 $m^3 s^{-1}$ (Loizeau and Dominik, 2000).

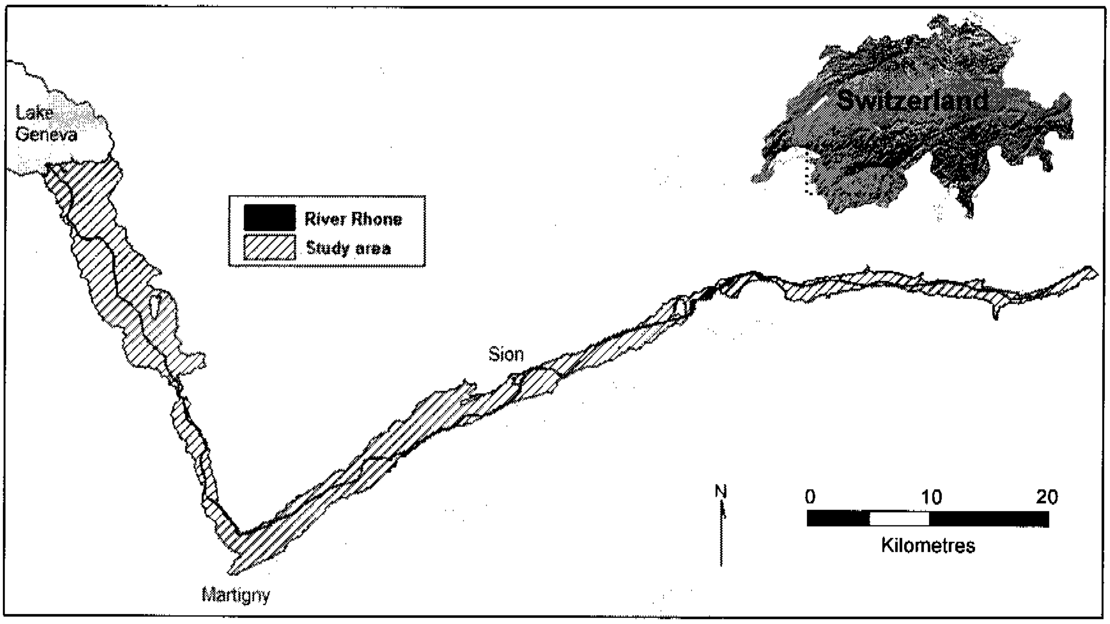

Figure 1: Location of the Rhone Plain in Switzerland. The current situation of the river Rhone is shown.

The valley was formed by the sweeping advance of the Rhone glacier at the beginning of the Quaternary Period (Département Fédéral de l'Intérieur, 1964). The river Rhone and its tributaries filled and formed the plain with their bed loads over the course of the centuries. The region has undergone fundamental compositional change over the past 150 years. The first systematic river correction, which was carried out in the late 19th century, almost completely destroyed the alluvial river-floodplain system, which covered 10.9% of the plain in 1850 and only 1.6% in 1900 (Zanini et al. in press). The braided active channel of the river Rhone was restricted to a single mainstem, resulting in a shoreline reduction of 150 km (45 %; Weber et al. submitted for publication).

Additional damage was caused to the ecological state of the plain after 1900 by the second river correction, the drainage of the plain, agriculture intensification and settlement development. Wetlands covered 6.1% of the plain in 1900 and only 1.3% in 2003. Furthermore, we observed a significant increase in urban area, in particular in the second half of the 20th century (1.4% in 1900 and 13.8% in 2003). Today, the Rhone plain constitutes a human-shaped landscape dominated by intensively exploited agricultural land (76.6%), in which the former natural ecosystems have almost completely disappeared (Zanini et al. in press).
Despite the two systematic corrections of the Rhone, the anticipated level of flood security was not entirely attained, and considerable damage was caused to the region by repeated and severe flood events in 1935, 1948, 1987 and 1993 (Wallis, 2000). As a result, it was decided in September 2000 to undertake a third correction of the river Rhone, with the help of which flood protection will be improved along a significant stretch of the river Rhone A simultaneous revaluation of ecological and socio-economic concerns is also planned (Wallis, 2000).

Historical sources

Based on the historical topographic maps mentioned in Table 1, 15 land cover classes (Table 2) were identified and digitized using ArcMap^{TM} 8.3 (ESRI). These classes represent the most detailed information obtainable from the interpretation of topographical maps.

Table 1: Cartographical sources used to reconstruct the former state of the Rhone Plain and to digitize land cover.

Period	Publication	General context	Name	Scale
1900	1882-1904	Reference state: after first river correction, still with significant wetland area and extensive agriculture.	Siegfried Map	1: 50 000
2003	2003	Current state: significant agriculture intensification and urban development.	Digital National Map of Switzerland	1: 50 000

Topographical maps of the plain were available for 1850 and were digitized in previous studies (Zanini et al. in press). Due to low spatial accuracy and a high level of incertitude with regard to land-cover identification arising from the measurement techniques and small map scale (1:100 000), it was not possible to use these data in this study.

Thus, we used the year 1900 as the reference state, which is also supported by ecological elements. Indeed, although the first Rhone river correction (1863-1876) produced dramatic ecological consequences with regard to the alluvial ecosystems, the overall ecological value of the plain and its conservation potential for biodiversity remained high. Flooding episodes were still frequent (Wallis, 2000) and several regions in the plain remained dominated by marsh, extensive agricultural practices and some unique ecosystems related to the Rhone alluvial system (e.g. sand dunes) were still present. Thus, as confirmed by several studies, overall species richness in the plain was still high in 1900 (Delarze, 1982; Desfayes, 1996; Farquet, 1924; Gams, 1916, 1927; Giugni, 1985; Rey et al., 1985).

Table 2: 15 land-cover classes digitized from the historical topographical sources for the years 1900 and 2003. A buffer corresponding to an estimation of real width was set around the linear topographical elements (tributary, canal, highway, railroad and road).

Land cover	Topographic map interpretation
Forest	Map signature for closed forest
Wetland	Map signature for wetland
Natural active channel river Rhone	Active side-arms or reaches of the river Rhone with intact floodplains, along one bank at least. According to Müller-Wenk (2004), intact floodplains are defined as areas between two arms of the river Rhone or outside of the furthermost arm carrying the signatures for gravel or open forest. Periodic inundations by the Rhone can be expected (absence of settlements and infrastructure). Both land and water area (mainstem, side-arms etc.) are included.
Rhone, canalized	Area occupied by river Rhone, but with none of the features mentioned under the floodplain, river Rhone.
Floodplain tributary	Alluvial area within the influence of a tributary of the Rhone.
Hill	Map signature for hill
Dune	Map signature for hill and cited in the literature as a dune (Farquet, 1924, Delarze pers. comm., 2004)
Urban area	Area including (1) >5 buildings within max. 100 m distance; (2) isolated, but large buildings covering approximately the surface of five smaller buildings (e.g. industry, factories)
Agricultural zone	Remaining area of the plain, essentially agricultural area
Tributary	Running water originating in the mountains and flowing into another river or canal or into Lake Geneva (width 6 m)
Canal	Running water originating in the plain or appearing at the surface after being piped (width 4 m)
Highways	Highway: width 30 m
Railways	Railroad: width 10 m
Roads	Roads of first and second class. Width: 4 m in 1900 and 6 m in 2003
Stagnant water	Stagnant water of natural or artificial origin

Data quality

The quality of the digitized data is crucial for any inter-year analyses. It may be influenced by the incertitude of cartographical sources, transformation through scanning and geo-referencing or the screen-digitations (Johnson, 1990; Kienast, 1993). Moreover, comparability between years is also influenced by data acquisition and generalization. In order to estimate the discrepancy between the 1900 and 2003 maps, the position of nine churches was compared. Churches were the topographical elements that could be identified most accurately at our working scale. Because the map scale is relatively small (1:50 000, 1 mm=50m!), we expected errors to occur, mainly due to geo-referencing. However, we observed a median discrepancy of only 24 m, which is sufficiently small to ensure limited errors in an inter-state comparison.

Estimation of the conservation value of the landscape

In terms of landscape patterns, historical topographical maps and aerial photographs provide an important basis for reference construction (Hohensinner et al., 2004; SER, 2004), which is available for many regions. In contrast, historical information on the biotic (e.g. species richness) and abiotic (water quality, soil type) state of ecosystems is generally difficult to obtain and is often extremely fragmented (for several examples see Egan and Howell, 2005a), especially if the study region is large (Van Diggelen et al., 2001). One way of dealing with this problem is to use expert knowledge. Expert knowledge offers significant support in conservation planning and ecological assessment when evaluation models based on empirical studies are not available (Balram et al., 2004; Hellier et al., 1999; Store and Kangas, 2001).

We selected three experts who are familiar with the ecological characteristics of the ecosystems in the Rhone plain and their evolution over the last century. These experts are biologists by profession, live in the study area and work as environmental consultants in private offices. We asked them to estimate the land-cover conservation value in relation to the number and rarity of potentially present species of each land-cover class identified in 1900 (reference state) and 2003 (actual state). The conservation value ranges from 0 (no conservation value) to 10 (maximum conservation value).

The conservation value of the entire plain (i.e. landscape conservation value) can be calculated using the following formula (equation 1):

$$C = \sum ps \qquad \text{(eq. 1)}$$

where C is the conservation value of the plain, p the proportion of the land-cover class and s its expert score. We used this weighted summation because (i) no synergic effect is considered in our evaluation and (ii) it is easy to explain and transparent (Janssen, 2001).

However, this result cannot be represented spatially. Therefore, we estimated the landscape conservation value in regular cells of 250 x 250 m in accordance with the aforementioned formula with p as the proportion of each land cover class in the cell. A total of 3,143 cells were delineated in the Rhone plain. As suggested by EUROSTAT (2000) and Chetelat (2005), cell size may affect the results. A small cell size accentuates diversity between cells and, conversely, large cells limit variability. The cell size used in our study was chosen as the best compromise between the scale of our study area, the incertitude regarding the boundary of the digitized land cover classes and the narrow shape of the Rhone plain.

Finally, the difference between the conservation value of the current state (2003) and reference state (1900) was computed for each cell. The result of this difference is consecutively termed ecological alteration. Ecological alteration ranges from -10 to 10. A negative value is interpreted as a loss in landscape conservation value as compared with reference state. Conversely, a positive value represents a gain in landscape conservation value. The closer to zero the value, the smaller are the changes.

The expert evaluations were compared separately for each temporal state using the non-parametric Friedman-Test for related samples and the Wilcoxon-Wilcox-Test (*a posteriori*). All of the spatial analyses were computed using Mapbasic 7.5 and Mapinfo 7.5 software (Mapinfo corporation ©, 1985-2003). SPSS 11.0 for Windows was used for the statistical analyses.

Defining rehabilitation potential

According to the proposed methodology, the reference conditions and the estimation of the ecological alteration of the landscape over time enable the identification of areas in which anthropogenic landscape transformations caused the most negative ecological effects. These areas also have the higher rehabilitation potential because the deviation from near-natural reference conditions is the most significant, therefore the potential ecological gains from rehabilitation projects are greater. In all rehabilitation projects, reference conditions play a central role in determining the rehabilitation potential of a site (Egan and Howell, 2005a; White and Walker, 1997). Moreover, variations over time and historical aspects are not the only factors that can support the estimation of rehabilitation potential. Current conditions also have to be considered because the objective of rehabilitation projects is not only to re-establish former conditions, but also to fundamentally improve the current ecological state (Hobbs and Harris, 2001). Hence, we assume that rehabilitation potential is the highest where (i) ecological alteration is the most negative and (ii) the conservation value is currently lower.

We proposed the use of a scale varying between 0 (no rehabilitation potential) and 5 (maximum rehabilitation potential). We considered a linear relationship between ecological alteration and rehabilitation potential (Figure 2), but assumed it to be 0 as long as the value of ecological alteration is positive. We also considered a linear relationship between conservation value in 2003 and rehabilitation potential. Finally, we calculated the mean between both rehabilitation potential values. Thus, for each cell in the landscape we obtained an estimation of its relative rehabilitation potential which varied between 0 and 5.

The proposed score scale and the linear function represent only one possible solution. The method is very flexible and can be quickly and easily adapted to the specific requests and strategies of project stakeholders and decision makers. This is an important point as the possibility of including various opinions and societal desires could influence the success of rehabilitation projects considerably (Hobbs and Harris, 2001).

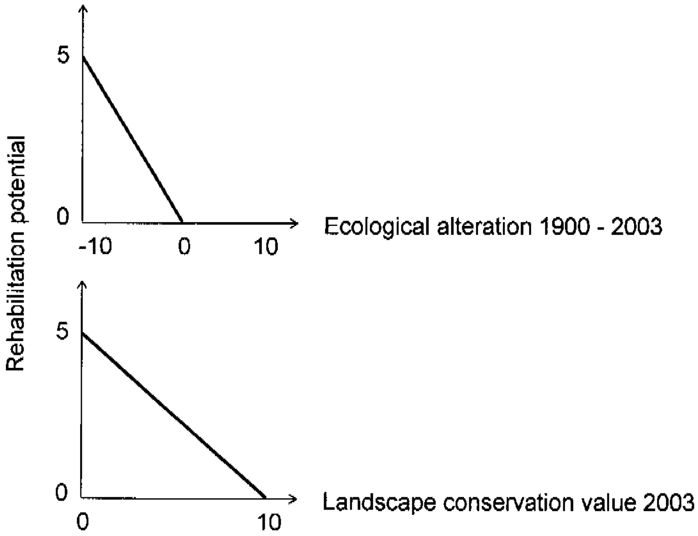

Figure 2: Criteria used for the estimation of rehabilitation potential based on spatial and temporal variation of conservation value. We assume a linear relationship between ecological alteration, conservation value and rehabilitation potential. Final rehabilitation potential is the mean between both estimations.

Results

The area of natural habitats (in particular wetlands and forest) decreased between 1900 and 2003 (Table 3). In addition, experts estimated that an intrinsic degradation of the ecological quality of land cover also occurred among years. This is particularly evident in the case of agricultural area, which received a mean conservation value score of 6.3 in 1900 and a mean score of only 2.3 in 2003 (Δ = -4).

In both states, the alluvial floodplain area of the river Rhone has the highest mean value for conservation (9.7 and 7.7 respectively). Dunes and wetlands also obtained high scores. As expected, urban areas, roads and highway have a lower value. Based on the mean land cover scores and the formula (1), the conservation value of the entire Rhone plain corresponds to 6.5 in 1900 and 2.5 in 2003 (Δ = -3.8).

This result is essentially dependent on the scores assigned to agricultural area which represents the predominant land cover in both temporal phases (78% of the Rhone plain area in 1900 and 73% in 2003).

Table 3: Land cover area (ha) and mean conservation value scores from the three expert evaluations (range is the difference between min and max expert score). Land covers are ranked in decreasing order for 1900. Conservation value ranges from 0 (no conservation value) to 10 (maximum conservation value). The Rhone plain conservation value is computed using the formula (1).

Land cover	Area [ha]			Mean conservation score				
	1900	2003	Δ area	1900	Range	2003	Range	Δ scores
Natural active channel Rhone	390	132	- 258	9.7	1	7.7	2	- 2.0
Dune	7	0	- 7	9.3	2	-	-	-
Wetland	1'439	283	- 1'156	9.0	2	7.3	1	- 1.7
Hill	40	25	- 15	8.7	1	7.3	4	- 1.4
Floodplain tributary	30	0	- 30	8.0	3	6.0	2	- 2.0
Canal	203	128	- 75	7.3	2	3.0	3	- 4.3
Stagnant water	9	154	+ 145	7.3	5	7.0	3	- 0.3
Forest	2'057	1228	- 829	7.0	3	6.7	3	- 0.3
Agricultural zone	18'610	17'474	- 1'136	6.3	4	2.3	1	- 4.0
Tributary	85	48	- 38	6.0	2	4.3	5	- 1.7
Rhone, canalized	568	573	+ 5.2	4.0	2	4.3	5	+ 0.3
Railways	132	156	+ 24	3.3	2	1.7	1	- 1.6
Urban Area	322	3'282	+ 2'960	3.0	3	2.3	3	- 0.7
Roads	83	257	+ 174	1.7	3	0.3	1	- 1.4
Highways	0	237	+ 237	-	-	1.7	2	-
Mean:				6.5	2.5	4.4	2.6	- 1.6
Conservation value Rhone plain:				6.5		2.7		-3.8

Relative agreement was observed in the score attribution by the three experts (EXP) (Pearson's correlation coefficient; EXP1-EXP2: $r = 0.76$; EXP2-EXP3: $r = 0.74$; EXP1-EXP3: $r = 0.76$) which would indicate that the experts share a common appreciation of the conservation value of the different land cover classes. However, significant differences existed between the experts for the evaluations of the 1900 state (Friedman-Test, $p = 0.027$). EXP 2 assigned significantly higher scores than EXP 1 and EXP 3 (Wilcoxon-Wilcox-Test, $p < 0.05$); there were no significant differences the scores allocated by EXP1 and EXP3. There were no significant differences in the expert evaluations for 2003 (Friedman-Test, $p = 0.736$).

The results of the ecological alteration and estimation of the rehabilitation potential are presented in Table 4. We assigned the cells in four classes of same size.

Table 4: Proportion [%] of the plain corresponding to different classes of ecological alteration and rehabilitation potential value. The views of all three experts are presented.

Ecological alteration classes	Description	EXP1	EXP2	EXP3
[-10, -5]	Very negative	44.0	39.6	3.9
[-5, 0]	Negative	56.0	59.4	93.3
[0, 5]	Positive	0	0.9	2.8
[5, 10]	Very positive	0	0	0
< 0	Negative and very negative	100	99.1	97.2
Rehabilitation potential classes				
[0, 1]	Very low	0	1.9	1.0
[1, 2]	Low	4.6	4.6	7.5
[2, 3]	Medium	39.3	18.4	85.2
[3, 4]	High	56.1	74.4	6.3
[4, 5]	Very high	0	0.8	0

According to the expert evaluations and computed land-cover changes, our method showed that ecological alteration in the Rhone plain was “negative” to “very negative” for more than 97% of the study area. For EXP3 ecological alteration was less negative than for the two other experts because the changes were classified as ecologically “very negative” for only 3.9% of the plain (as compared with 44.0% and 39.6% for EXP1 and EXP2, respectively). Moreover, positive ecological alteration, i.e. an increased conservation value, was found for only two experts and for an extremely limited area of the plain (0.9% for EXP2 and 2.8% for the EXP3).

In order to identify spatially where ecological alteration is “very negative” and where the results for three experts are consensual, we superimposed the three maps of ecological alteration and selected those cells for which all three experts had the same class value. Figure 3 shows the hot spots of “very negative” ecological alteration for a section of the study area. The areas corresponding to “high” or “very high” rehabilitation potential were computed using the same methodological approach and are also illustrated for the same section of the plain (Figure 3). These hot spots generally overlapped. Positive ecological changes are not mapped as the results of three experts are not spatially consensual.

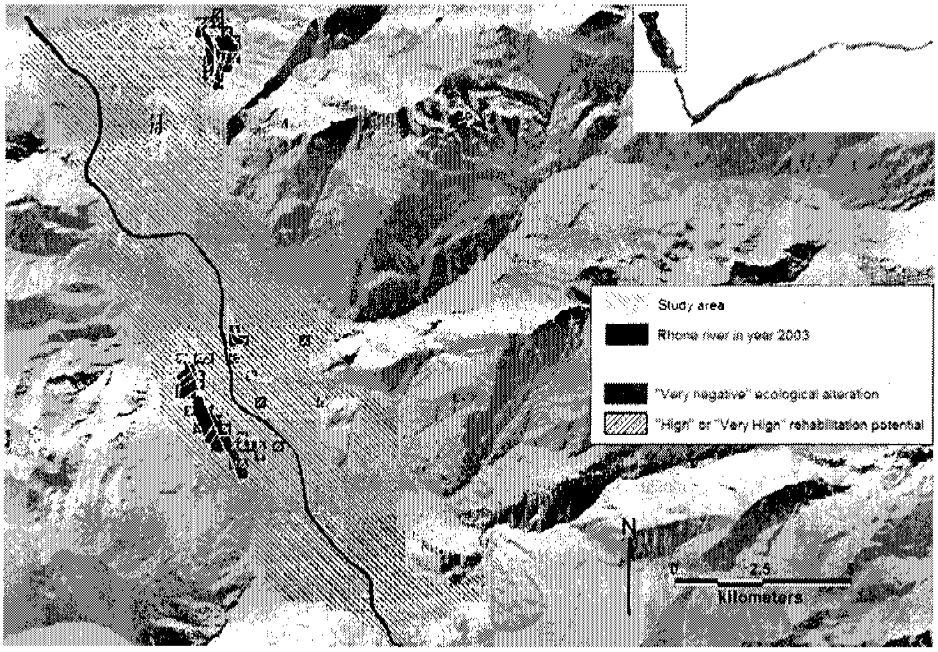

Figure 3: Section of the Rhone plain (the Chablais) close to the mouth into Lake Geneva. The map shows the hot spots of "very negative" ecological alteration ([-10, -5]) and the hot spots of "high" or "very high" rehabilitation potential ([3, 5]), on which all three experts agreed. The white boundaries indicate the wetland perimeter in 1900 (MNT25, © Swisstopo, 1995).

Discussion

In this study, we presented a method for the quantification of the spatial and temporal variation in the nature conservation value of the landscape. Our approach enabled the localization of hot spots where land cover changes were most negative or positive from an ecological point of view. Our method is simple to understand, appropriate for different spatial scales and flexible because land-cover classes, cell size and expert scores can be modified easily and quickly. Rather than computing a mean value, we also identified consensus among experts so as to respect each specific expert opinion, which is very important for the communication and acceptance of results (Maystre and Bolliger, 1999; Pictet, 1996). Finally, due to its standardized and transparent method which indicates where the rehabilitation potential of the landscape is highest when an historical state is used as reference, our approach could have important implications for landscape planning.

In this study, we presented and discussed the implications for rehabilitation projects. However, we also noted that the map produced, which provides information about the current conservation value of the landscape, could be very useful for the definition of conservation strategies, e.g. for the selection of area to be protected.

As is commonly acknowledged, the establishment of the reference state is a very difficult and time-consuming task and past data provide only snapshots of system parameters (Hobbs and Harris, 2001). Problems arise both due to the lack of historical information and the difficulty in defining what a reference state is (Egan and Howell, 2005b). If the reference constitutes a landscape that is entirely devoid of human influence, it would be necessary to obtain pre-historic information because human impact, e.g. in the Rhone plain, is documented back to the Neolithic period (Département fédéral de l'intérieur, 1964), and this would be a difficult task. Around 1850, the river Rhone was still in a relatively natural state with high-quality longitudinal and lateral connectivity (Weber et al. submitted for publication). The fish-species richness of the river Rhone was also appreciably higher at the end of 19th century (19 species) than today (only 2 species; see Weber et al. submitted for publication). However, at this time landscape was also already partially shaped by human activity (agriculture, farming practices and forestry) (Bender, 2001; Farquet, 1924; Kuonen, 1993). Several factors may influence the choice of the reference state, but, as suggested by Bradshaw (1997), the reference system is not necessarily intended to describe only the former state unaffected by humans. The reference must also be defined in accordance with the goals of the project, the availability of data and stakeholder acceptance. In our study, the 1900 state fulfilled these considerations.

For the Rhone plain we found that significant changes in landscape composition occurred over the past century with generally negative consequences for its ecological state. With the exception of stagnant water, which increased in area from 9 ha in 1900 to 154 ha in 2003 (+145 ha), the area of all natural habitats (see Table 3) decreased from 1900. The increase of stagnant water area is related to the development of gravel mining during the last century. This activity multiplies the number and size of ponds and associated pioneer habitats. Such ecosystems created by human activity are of particular importance for many endangered species in the region, for example the yellow bellied toad (*Bombina variegata;* Grossenbacher, 1988).

However, because of its small size and the generally extreme degradation of the ecological state of the study area in 2003 the effect of newly created stagnant water ponds on conservation value of the landscape was limited.

The hot spots which experienced the greatest negative ecological alteration are essentially located in the former wetland areas, which were still significant in size at the beginning of the last century (1439 ha, 6% of the plain), but have now almost entirely disappeared (283 ha, 1% of the plain). Wetland conversion was essentially due to agriculture rationalization and intensification. Indeed, previous studies revealed that 81% of the wetlands that existed in the plain in 1900 had been converted into agricultural area by 2003 and almost 7% had been converted into urban area (Zanini et al. in press). The ecological importance of wetland for biodiversity conservation and the dramatic consequences arising their reduction have been documented for the plain. For example, between 1882 and 1982, at least 65 plant species associated with marsh area disappeared in the Chablais area on the right bank of the river Rhone (Figure 3) (Delarze et al., 1982). In the same region on the left bank of the Rhone, 116 plant species disappeared between 1850 and 1985 (Giugni, 1985) and 98 plant species probably became extinct in the regions close to the mouth of the Rhone into Lake Geneva (Morel, 1985). Other species associated with wetland ecosystems also declined in the region. Among these, amphibians are emblematic of species that are sensitive to habitat degradation (Dodd and Smith, 2003). According to the Swiss Red List, eight out of twelve species are endangered, vulnerable to or threatened with extinction within the study area, (Schmidt and Zumbach, 2005). Many studies have noted the worrying threat to amphibian populations in the Rhone plain and have identified some possible causes such as the destruction of breeding ponds, road mortality, natural succession of the vegetation and expansion of the invasive species *R. ridibunda* (Berthoud, 1976; ECOTOEC, 1996; Farquet, 1924; Grossenbacher, 1988; Jordan et Rey, 1973; Marchesi, 1999; Praz, 1983; Rey et al., 1985). Other possible causes exist, such as climate change, increased UV-B exposure, agrochemicals and chemical pollution, human exploitation and disease (Alford and Richards, 1999; Blaustein and Kiesecker, 2002; Kiesecker et al., 2001). However, in the context of the Rhone plain, the dramatic habitat destruction (in particular of wetlands) and fragmentation and their direct and indirect consequences are probably the main drivers of amphibian decline.

In addition to changes in landscape composition, which were considered in the present study, changes in landscape configuration have also triggered a reduction in the suitability of landscape for species persistence (Zanini et al. in press). The number of forest patches increased by a factor of 2.6 (175%) from 1900 while their average size decreased by 78%. The number of wetland patches and their average size also decreased (by 66% and 40% respectively). Conversely, the number of urban area patches increased by 122% and their average size by 344%. This confirms the increasing of urbanization process with growing agglomerations and newly-created urban areas. As a result, because they were often isolated and stressed by the surrounding environment, the remaining natural areas offer fewer suitable living conditions for the species, (Zanini et al. in press). Thus, it is our view that in order to obtain a more realistic ecological assessment, the approach proposed in this paper could be improved by considering supplementary criteria such as patch rarity, diversity, isolation, naturalness and exposure to disturbances (Geneletti, 2002, 2004a, 2004b; Lee et al., 1999, 2001, 2002; Lesslie et al., 1988; Morgules and Usher, 1981; Spellenberg, 1992). However, in the case of broad-scale landscape evaluation like this research, the utilization of multiple criteria may be inadequate due to (i) the complexity of the system under study (15 different land cover classes) and (ii) the subsequent potential difficulty in communicating the results to stakeholders (Jansenn, 2001).

Assessment based on expert knowledge is usually used when it is impossible to carry out an objective evaluation due to the lack of data. In principle, by including expert knowledge, we fill the gaps in data with the subjectivity of the experts. In order to limit errors in evaluation and to approximate the conservation value of the landscape as accurately as possible, in our study we were very strict in the selection of experts. The evaluation was a difficult exercise, in particular for 1900 as very little descriptive information was available on the state of ecosystems, species richness and human impacts. This incertitude resulted in significant differences between the expert evaluations carried out for 1900. In contrast, there were no significant differences between the evaluations for 2003, which would indicate the existence of a consensual perception of the conservation value of the landscape and validate the assessment of its current state. Thus, in order to reduce the discrepancies between the expert assessments, only the hot spots with very negative ecological alteration identified by all the experts were discussed.

In our study we estimated rehabilitation potential spatially and proposed that it be used to locate priority areas for rehabilitation. Decisions in the area of rehabilitation (or restoration) ecology could be driven and supported by historical information (Egan and Howell, 2005a). However, we agree that historical information and rehabilitation potential is only one element that can be considered in the planning of rehabilitation and restoration projects. Lindenmayer et al. (2002) stressed that the complexity of ecological problems at landscape scale suggests that there may not be just one straightforward way for setting targets for urgently needed restoration projects. Instead, it will be important to adopt a risk-spreading approach which involves the implementation of a wide range of strategies for landscape restoration. Moreover, to succeed, restoration activities not only need to be based on sound ecological principles and information, they must also economically feasible and practically achievable (Hobbs and Harris, 2001). This implies that political acceptability often plays a more important role in the setting of priorities and choice of options than any rational process (Hobbs and Harris, 2001). It is our belief that *in situ* historical elements are essential components of the decision making process in this context. As citizens, resource managers, and policy makers become more familiar with well-validated and locally-generated pictures of landscape history, a shared understanding of present conditions and potential future scenarios becomes more possible, and a common vision of the future can emerge (Antrop, 2005, Grossinger, 2005).

Conclusion

The main contribution made by this study is its development of a feasible method for monitoring both spatial and temporal changes in nature conservation value and the identification of hot spots of landscape rehabilitation potential. Our approach is based on the historical reference state (1900) and on expert assessment of landscape conservation value. Due to the complex structure of fragmented and heterogeneous landscapes, significant assumptions and simplifications were made. However, simplification produced comprehensible results which can be quickly calculated and easily explained and understood (Jansenn, 2001; Young and Jarvis, 2001).

As suggested by Hobbs and Harris (2001), assessment processes carried out in the context of restoration project can be complicated and expensive, and if they are too complicated or expensive, they will not be carried out.

Historical references are only one of the sources that may be considered in the planning of nature conservation measures. However, the use of a reference state as target for a rehabilitation project is an important step, in particular if current landscape conditions show dramatic degradation and information is required regarding the pristine state (e.g. Jungwirth et al., 2002). The proposed methodology represents an additional helpful element for landscape planning. It could play an important role in facilitating communication in participation processes and provide a useful tool for decision makers involved in projects at landscape level and for sustainable land-use policy which is an important element of the Swiss Landscape Concept (OFEFP, 1998; Stremlow et al., 2003).

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CHAPTER 4

Habitat diversity and fish assemblage structure in local river widenings: a case study on a Swiss river

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Abstract

Habitat rehabilitation projects in running waters are often conducted on a local scale and their effect on the local biota or the larger river system is unclear. We investigated habitat availability and fish assemblage structure in three local river widenings, completed 3 to 14 years ago, and five adjacent canalized reaches on the river Thur, a seventh-order river in Switzerland. To account for seasonal variability, surveys were repeated in winter and summer 2005. Results were compared with historical pre-disturbance data to evaluate whether the current abiotic and biotic conditions in the study reaches have attained near-natural levels.

Hydro-physical habitat diversity (depth, flow velocity, cover availability) was considerably greater in the two longer widenings (> 900 m long) than in the canalized reaches and in the shortest widening (300 m long), with higher proportions of shallow or deep areas of different flow velocities. However, the comparison of current and historical near-natural shoreline lengths indicated that the current geomorphological complexity is still considerably impaired in all reaches.

No overall significant relationship was found between the reach type (canalized or rehabilitated) and the number of species or the total fish abundance which were strongly correlated with the availability of suitable cover and moderate flow velocity. However, highest winter abundances were observed in deep, well-structured backwaters of the rehabilitated reaches, documenting their significance as wintering habitats. Assemblage structure and composition were similar in canalized and rehabilitated reaches. Compared to the historical data, however, fewer and different dominant species were found, and guild composition changed towards a higher representation of generalists and tolerant species.

Keywords: River rehabilitation, canalization, fish assemblage, habitat

Introduction

Riverine fish assemblages are structured by diverse habitats (Cunjak 1996; Matthews 1998; Schlosser 1991). Therefore, most rehabilitation measures attempt to re-establish the structural complexity lost by previous anthropogenic impacts (Gore and others 1995), such as canalization or stream clearing. In many rehabilitation projects habitat structures are actively created, for example, through the placement of boulders (Lepori and others 2005), large woody debris (Hilderbrand and others 1997) or spawning gravel (Iversen and others 1993). Most of these measures emphasise individual, usually salmonid species (Frissel and Ralph 1998). Several studies have documented a limited durability of artificial habitat structures (Frissell and Nawa 1992; Linlokken 1997; Roni and others 2002), particularly in streams with high peak flow or high sediment load. In recent years, rehabilitation has therefore increasingly focused on the recreation or maintenance of ecosystem processes or functions (Angermeier and Karr 1994; Frissel and Ralph 1998; Muhar and Jungwirth 1998). In functioning ecosystems, site-specific habitats are naturally formed and connected (Beechie and Bolton 1999; Roni and others 2002), over various spatial and temporal scales. Patches are continually destroyed and recreated by spatio-temporal variations in processes. In this way, a shifting mosaic of diverse habitats is formed, which fulfils the requirements of different fish species and age classes (Bormann and Likens 1979), depending on temporal and spatial scale.

The re-establishment of natural riverine dynamics is one of the objectives of local river widening, a rehabilitation measure mainly used hitherto in canalized rivers in Austria, Germany and Switzerland. The river bed is significantly widened along a particular stretch through the removal of the embankments and the setback of the flood levees (Rohde and others 2005b). Channel migration and braiding are enabled within the widened reach, leading to greater structural and hydraulic heterogeneity (Peter and others 2005; Woolsey and others submitted). Widenings are particularly appropriate for the rehabilitation of formerly braided rivers with intact or little impaired bed load. A technical aim of this measure is to halt river bed erosion (Peter and others 2005).

System-wide enhancement is not possible for most canalized rivers, and widenings are often conducted as local measures. However, their effect on the local fish fauna or the larger river system has not been documented.

The following questions need to be addressed: (i) is the potential physical and biological response restricted to the restructured area or are positive consequences also observable in adjacent reaches and (ii) can rehabilitated river reaches serve as colonization sources for canalized ones?

In this case study we compared the habitat availability and fish assemblage structure in three local river widenings, completed 3 to 14 years ago, and five adjacent canalized reaches on the river Thur, a seventh-order river (Pfaundler 2005) in north-eastern Switzerland. Special consideration was given to the spatial setting of each sampling station in the interpretation of the fish data, i.e. its location in the system and its distance from a near-natural river reach such as a widening. To account for seasonal variability, surveys were repeated in winter and summer 2005. Results were compared with historical pre-disturbance data to evaluate whether the abiotic and biotic conditions of rehabilitated reaches have attained near-natural levels. We derived practical implications for the design of local river widenings from our findings.

Material and Methods

Study site

The river Thur, a tributary of the river Rhine (High Rhine), is located in the north-eastern part of Switzerland (Figure 1). Along its 127 km length it drains a catchment of 1,750 km^2 and overcomes an altitude difference of 1,150 m. The average annual discharge close to the mouth is 47 m^3 / s. The flow regime is flushy: strong rainfall in the catchment gives rise to a very rapidly increasing water level up to 723 m^3 / s (HQ_5, five year flood) or 817 m^3 / s (HQ_{10}).

As a result of a large-scale modification in the late 19th century, the Thur is mainly canalized today and most of the floodplains are disconnected from the river. The longitudinal connectivity is disrupted by seven dams and numerous weirs (Schager and Peter 2005). Only the lowest 36 river kilometres provide free passage for migratory fish (Schager and Peter 2005).

After repeated severe flood events, it was decided to carry out a second large-scale river modification in the 1970s with the main focus on flood protection. Swiss law also requires the simultaneous enhancement of ecological and landscape conditions. As a result, about 15 river widenings have also been completed over the past 16 years.

These widenings differ considerably in their spatial dimension and design: based on the accumulation of experience, longer and wider river widenings have been implemented and the structuring of the active channel has been increasingly left to the dynamic of the river itself.

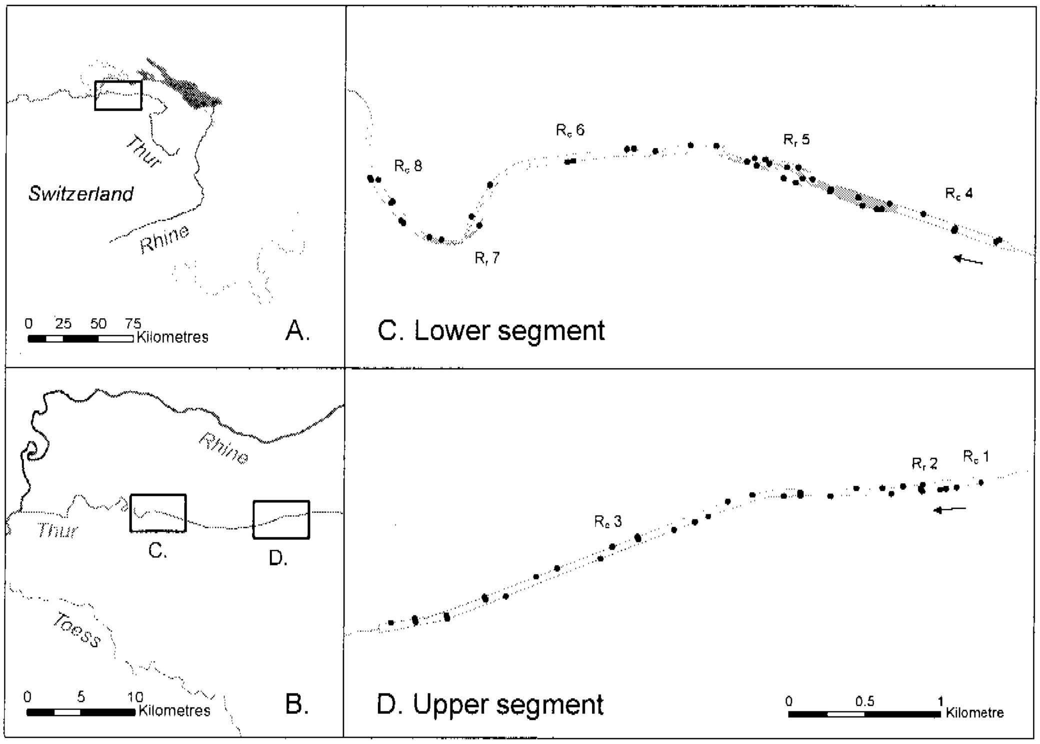

Figure 1: Location and dimension of the study segments and reaches (R). The wetted areas in inset B and C were determined in winter 2005 by means of a dGPS (TSC1; Trimble, Sunnyvale, CA, U.S.A.). Rehabilitated reaches (R_r) are shown in grey, canalized reaches (R_c) in white. Points indicate sampling stations.

Study design

We concentrated on the three widenings (Figure 1) where the channel was considerably widened (> factor 1.5; Table 1) and the five canalized reaches adjacent to them. The eight reaches can be grouped into an upper (river km 28.0 - 32.2) and a lower river segment (river km 15.5 - 20.6). Both segments are located at the transition from barbel to grayling zone (Schager and Peter 2005).

Area-wide habitat mapping was carried out in all eight reaches and 76 sampling stations were studied by means of electrofishing.

Sampling and mapping took place in winter (February) and summer 2005 (June and August) under comparable discharge conditions of about 19 m^3 / s. This corresponds approximately to a Q_{247}, i.e. a discharge that is reached or exceeded for 247 days on average per year.

Table 1: Characteristics of canalized and rehabilitated reaches of the river Thur. * Factor by which the canalized channel was widened (Hunzinger 1998; Rohde and others 2005a).

Reach	Segment	Reach type (year of completion, dimension*)	Length (km)	Number of stations for electrofishing
R_c 1	Upper	canalized	0.3	2
R_r 2	Upper	rehabilitated by widening (1991; 1.8)	0.3	6
R_c 3	Upper	canalized	3.6	26
R_c 4	Lower	canalized	0.8	5
R_r 5	Lower	rehabilitated by widening (2002; 2.2)	1.9	23
R_c 6	Lower	canalized	0.5	2
R_r 7	Lower	rehabilitated by widening (1992; 1.7)	0.9	5
R_c 8	Lower	canalized	0.9	7

Habitat mapping (reach scale)

Mesohabitats were mapped in the field in all reaches. A mesohabitat is defined as a quasi-discrete area which is homogenous with respect to the factors of water depth, current velocity and availability of fish cover. We divided each factor into two or three categories (Table 2) which combined to form a total of 18 possible habitat types. Categories were visually estimated in the field.

Table 2: Categories used for habitat mapping.

Variable		Level 1	Level 2	Level 3
Depth	[cm]	≤ 30	30 – 110	> 110
Flow velocity	[cm / s]	≤ 10	10 – 50	> 50
Cover availability*	[%]	≤ 5	> 5	

* Cover was defined as the area providing shelter from predators and high current velocities. Both overhead cover and slow water areas behind submerged objects were considered in accordance with Peter (1992).

The mapping of all reaches was conducted within four days. Whenever possible, mesohabitats were delineated by walking along them carrying the receiver of a differential Global Positioning System (TSC1; Trimble, Sunnyvale, CA, U.S.A.). Position was recorded automatically every two seconds.

Non-wadeable instream areas were mapped from the shore by visually estimating patch dimension and recording geographical reference points. Field data were later corrected using records from a stationary reference receiver (Zurich, Switzerland) and resulted in a measurement precision of ≤ 0.3 m.

Spatial data were converted into habitat maps using ARCMap 8.3 (ESRI). Visually estimated habitat patches were added manually through digitization. The total and relative area of each habitat type were determined for each reach and the habitat diversity (using Shannon's index of diversity; Krebs 1989), the number of habitat types and the evenness (Shannon's evenness; Krebs 1989) were calculated.

Electrofishing and habitat characterization (station scale)

a) Selection of sampling stations

Sampling stations were distributed randomly using ArcMap 8.3 (ESRI). The occurrence and spatial proportion of the different mesohabitat types were considered (proportional sampling), e.g. by inclusion of rare and underrepresented mesohabitat types. A total of 76 stations were sampled (Table 1).

b) Electrofishing

The selected sampling stations were studied by means of semi-quantitative electrofishing (one pass, no block nets). At each station, a strip of at least 25 m in length and of an area of about 100 m^2 was fished. Due to their usually smaller dimension, specific mesohabitat structures, such as backwaters or isolated pools, were fished over the entire surface.

A stationary electroshocker was used in most cases (EFKO, 8 kW, 150-300/ 300-600 V). Two stations which were difficult to access were fished using backpack equipment (EFKO, 3 kW, 150-300/ 300-600 V). Non-wadeable stations, such as deep backwaters, were sampled from an inflatable boat tied ashore.

Captured fish were handled in accordance with a standardized procedure including controlled conditioning and anaesthesia with clove oil (Hänseler AG, Herisau, Switzerland; 0.5 mL diluted in 9.5 mL alcohol added to 15 L water). Because of difficulties in catch and determination, young-of-the-year fish hatched in 2005 were excluded from the survey. Fish species, total length (±1 mm) and presence and type of any anomalies were determined. All fish were released along the fished stretch after recovery.

The number of species and the fish abundance in total and per species [individuals / 100 m^2] were calculated for each station.

c) Habitat characterization of the sampling stations

In order to specifically characterize local conditions, habitat measurements were recorded at each station after electrofishing. Mean water depth was calculated from fifteen to twenty measurements made in regular intervals along the fished stretch, and current velocity was visually estimated (Table 2). The dominant substratum was assigned to one of nine classes using a modified Wentworth scale (Cummins 1962). Presence and type of suitable fish cover were determined visually (%; Table 2). In order to consider the proximity of potential sources of colonizers, the distance to the closest near-natural river reach upstream and downstream ("Dist up" and "Dist down" variables, respectively) was determined for each station. The position of each station in the river Thur was described in terms of the distance from the mouth into river Rhine ("Dist mouth" variable). The reach membership of each sampling station was defined by the variable "reach number".

Analysis

Habitat data (reach scale): due to the small number of replicates (three rehabilitated reaches and five canalized reaches) habitat data could only be analysed graphically.

Fish data: for each reach type (canalized or rehabilitated), taxonomic richness was determined. As species numbers are influenced by the number of individuals captured per station, they must be standardized to a common number of individuals. Using rarefaction functions (Estimate S software program; Colwell 2005), the expected species number, given n individuals, is calculated analytically and not by iterative re-sampling (Gotelli and Colwell 2001).

Abiotic variables and fish abundances were compared using a general linear model (GLM; SPSS 13.0 for Windows). All abiotic variables were introduced into the model as covariates, except for the reach number. As it represents a nominal measure, it was treated as a fixed factor, nested in the variable reach type.

Prior to the analysis, all interval-scaled data were log-transformed [log10 (1+x)] to homogenise variances of residuals. Analysis was carried out separately for the two seasons.

A standard Bonferroni correction was applied to compensate for the increased likelihood of finding a significant result when calculating multiple –i.e. three– statistical models with the abundance data (total abundance, abundance of chub, and abundance of spirlin). The common significance level $p < 0.05$ was divided by the number of tests performed, resulting in a significance level of $p < 0.017$.

Reference conditions

Rehabilitation measures tend to improve ecosystem functions and structures towards more natural conditions (Bradshaw 1996). Comparisons with a guiding image or *Leitbild* describing the near-natural reference state are important for the evaluation of the degree of this approximation and the presence of ongoing shortcomings. Historical information, such as maps or documentary sources, offers an alternative when natural rivers are lacking as current references (Hohensinner and others 2004; Jungwirth and others 2002; Kondolf and Larson 1995).

Indicators that characterize relevant ecosystem elements are required for the comparison between current and reference conditions. We chose three indicators (Table 3) to reflect aquatic habitat diversity along the shoreline ecotone and the structure of the fish assemblage.

Table 3: Indicators and related reference information for the determination of the degree of naturalness of habitats and fish assemblage structure in canalized and rehabilitated reaches in the river Thur.

Indicator:	Shoreline length	Number and abundance of fish species	Ecological guilds of fish
Measurement criteria:	Ratio of shoreline length to river length [km / km]	- Total abundance of fish - Absence of native species - Presence of exotic species - Dominance structure	- Number of guilds - Strength of guilds
Reference information:	Historical map (Wild Map, 1851, 1:25'000) digitized by using ArcMap 8.3 (ESRI).	Historical fish inventory of the river Thur (Wehrli 1892) providing information concerning species occurrence, distribution and abundance (see Table 5).	
Evaluation criteria:	Ratio of current s. length to reference s. length. The minimum value of 2 (km / km) is subtracted from both values prior to the calculation.	Qualitative evaluation in words.	

The indicators were determined in accordance with the Swiss guidelines for success evaluation in river rehabilitation projects (Woolsey and others submitted; Woolsey and others 2005). Fish indicators in these guidelines are mainly based on Schmutz and others (2000). The degree of naturalness was either expressed as the percentage from the reference value (shoreline length) or in words (fish indicators).

The current high resolution shorelines recorded in the field by dGPS were simplified using the simplify line function in ArcMap 8.3 (bend-simplify, 10 m) to render them comparable with the lower resolution historic maps. This led to minor reductions in shoreline lengths of between 1 % and 4 %.

Results

Habitat data

Habitat diversity

Regarding the different diversity measures the ranges of values for canalized and rehabilitated reaches overlapped (Figure 2). The maximal values for all three variables, however, were for both seasons attained in rehabilitated reaches. Among the rehabilitated reaches, R_r 2 performed worst whereas R_r 5 and R_r 7 behaved similarly.

For both seasons, there was a highly significant correlation between habitat diversity and current shoreline lengths (Pearson correlation coefficient r_{winter} = 0.862, p_{winter} = 0.006; r_{summer} = 0.795, p_{summer} = 0.018).

Comparison with reference conditions

The historical shoreline lengths varied considerably between the different reaches (Table 4), but were in general longer than today. The current shoreline lengths in all the canalized reaches almost reached the minimum of 2 km / km. With the exception of R_c 1, where both reference and current values were close to the absolute minimum of 2 km / km, this corresponds to a proportion of less than 10 % of the reference. The rehabilitated reaches showed consistently longer shorelines than the canalized reaches and thus achieved generally higher percentages.

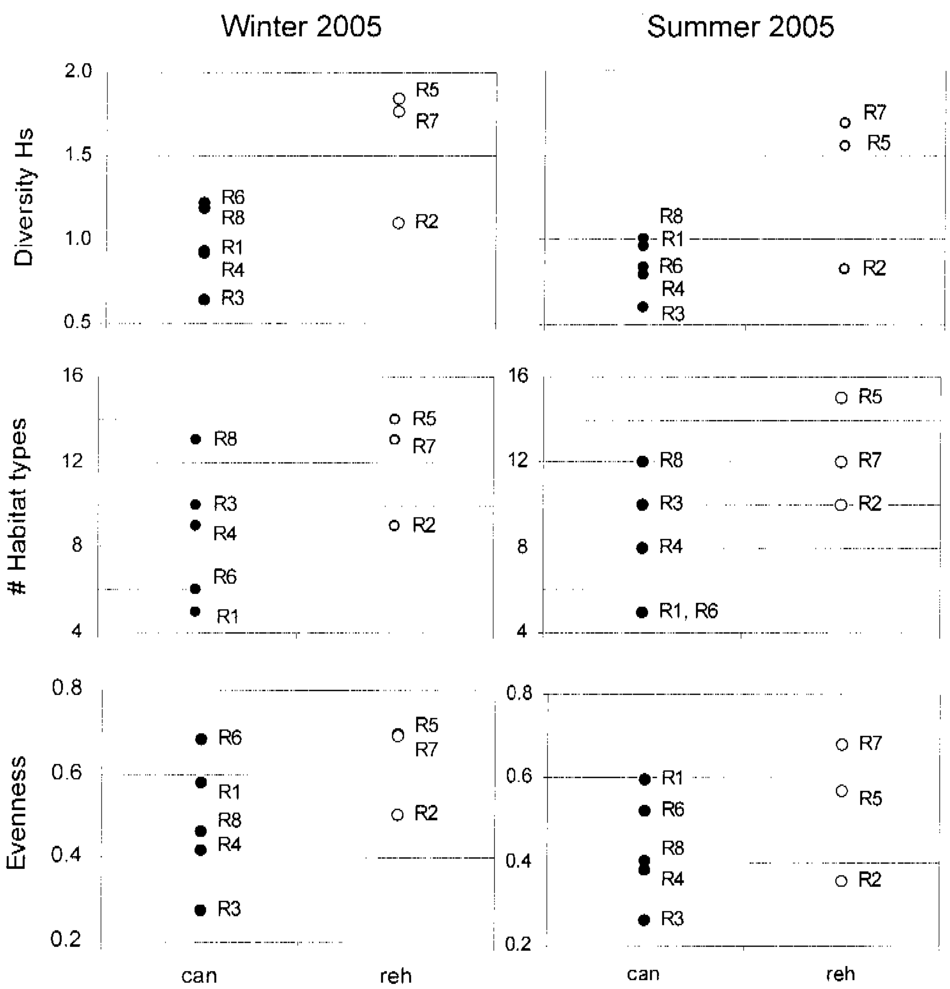

Figure 2: Characteristics of habitat data in canalized (can) and rehabilitated (reh) reaches in the river Thur.

Table 4: Shoreline length in the canalized and rehabilitated reaches in river Thur. For 2005, the mean from winter and summer mapping is shown. * Ratio of current shoreline length to reference shoreline length. The minimum value of 2 (km / km) is subtracted from both values prior to the calculation.

	Canalized					Rehabilitated		
	R_c 1	R_c 3	R_c 4	R_c 6	R_c 8	R_r 2	R_r 5	R_r 7
1851 (reference)	2.02	3.14	4.96	3.63	2.23	2.01	4.47	3.27
2005 (mean)	2.01	2.02	2.07	2.00	2.02	2.15	2.80	2.42
Percentage from reference [%]*	83	2	2	0	8	> 200	32	32

Fish data

Species diversity and population structure

A total of 4,149 individuals belonging to 20 species were caught over both sampling campaigns (Table 5). In winter, 2,576 individuals originating from 14 species were caught. The spirlin, *Alburnoides bipunctatus*, was the numerically dominant winter species in both reach types, followed by the European chub, *Leuciscus cephalus*. 1,573 individuals belonging to 18 species were observed in the summer sampling. The European chub was the numerically most abundant species in both canalized and rehabilitated reaches, followed by the spirlin. Young animals dominated in most species (Table 5), whereas adult fish were generally underrepresented compared to an intact population structure.

In both seasons, no difference was found in the standardized species numbers of the canalised and rehabilitated reaches (Figure 3), as demonstrated by the overlapping confidence intervals of the rarefaction curves.

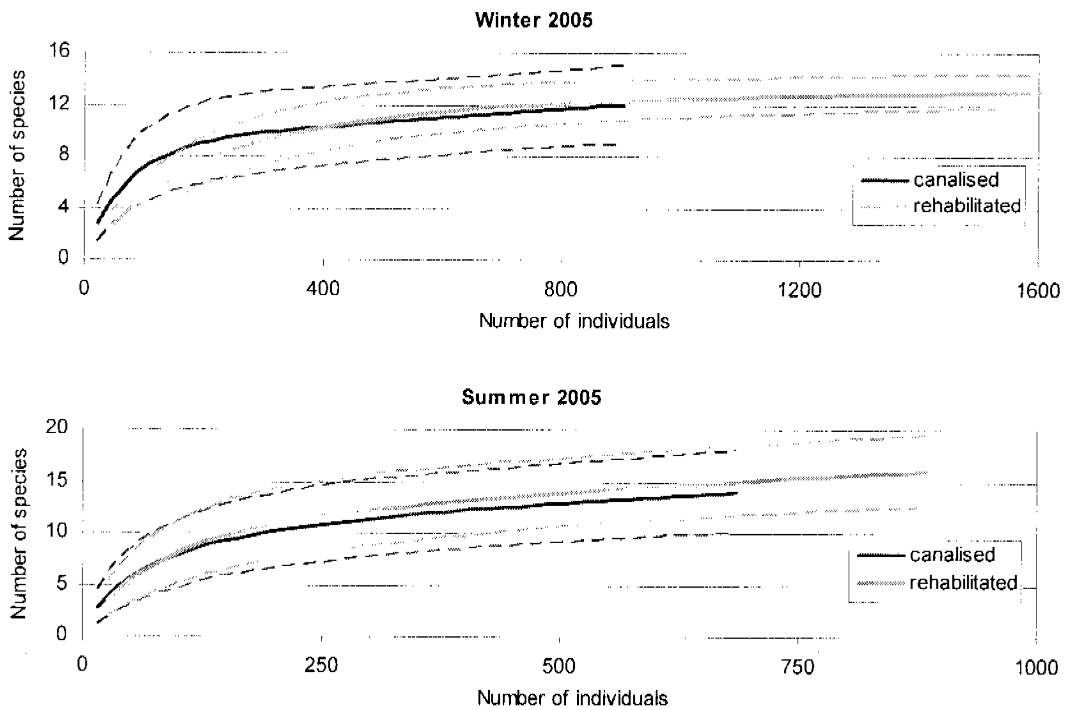

Figure 3: Individual-based rarefaction curves demonstrating the taxonomic richness in canalized and rehabilitated river reaches in winter and summer. Dashed lines indicate 95 % confidence intervals.

Table 5: Species represented in the seasonal catches (relative abundances) and their historical occurrence in the river Thur. A distinction is made between canalized (can) and rehabilitated (reh) reaches. * Indeterminable cyprinids. Actual occurrence: population structure is indicated as follows: Y = predominantly young fish (0^+, 1^+), A = predominantly adult fish (> 1^+), S = individuals from several age classes. Historical occurrence: based on a detailed description (Wehrli 1892), we classified each species' occurrence as very high, high, medium or low. Species that are not mentioned in the historical source are indicated as follows: [a)] native species in Switzerland, [b)] non-native species in Switzerland.

Species	Actual occurrence (%)				Historical occurrence
	Winter		Summer		
	can	reh	can	reh	
Alburnoides bipunctatus	70.9 Y	39.5 S	22.4 Y	9.6 Y	very high
Alburnus alburnus				0.1 Y	low
Anguilla anguilla	3.6 A	1.0 A	17.1 A	3.5 A	medium
Barbatula barbatula	2.1 Y	0.5 Y	1.9 Y	6.9 Y	high
Barbus barbus	4.0 Y	0.5 S	20.8 Y	0.8 Y	very high
Carassius spp.			0.1 A		b)
Chondrostoma nasus	0.1 A	0.3 Y		0.6 S	very high
Esox lucius		0.1 A			high
Gasterosteus auculeatus		0.1 Y		0.1 Y	a)
Gobio gobio	1.3 Y	1.8 Y	4.6 Y	3.2 Y	high
Leuciscus cephalus	8.2 Y	33.5 S	27.1 S	61.0 S	high
Leuciscus leuciscus	0.1 Y	0.7 Y	1.9 S	2.8 Y	low
Leuciscus souffia souffia	2.7 Y	12.4 Y	0.7 Y	2.3 Y	very high
Oncorhynchus mykiss	0.1 A				b)
Phoxinus phoxinus	4.3 Y	6.1 Y	1.9 Y	5.3 Y	high
Pseudorasbora parva			0.6 Y	3.2 Y	b)
Rutilus rutilus			0.1 Y		low
Salmo trutta fario	0.9 S	0.7 S	0.1 A	0.1 A	medium
Scardinius erythrophthalmus			0.1 Y	0.1 Y	low
Tinca tinca				0.3 S	restricted to lakes
*Cyp. spp.**	1.7 Y	2.8 Y	0.4 Y	0.1 Y	
Thymallus thymallus					low
Lampetra fluviatilis					low (seasonal occurrence)
Lampetra planeri					medium
Perca fluviatilis					low
Cottus gobio					high
Salmo salar					low (seasonal occurrence)
Total number of individuals	920	1'656	689	884	

Fish abundances

In winter, median total abundance was highest in R_c 6 (Figure 4). Extreme winter values were reached in R_r 5 and R_r 2. At 25 stations, no fish were caught in winter. Such stations could be found in almost every reach; 12 of them were located in R_r 5 leading to a winter median of 0.

Fish abundances in summer were clearly below those observed in winter (Figure 4). Median total abundances were highest in R_r 5, whereas maximum total abundances were found in R_r 5 and R_c 3. At six stations, we found no fish in summer. In winter, no fish have already been observed at these stations.

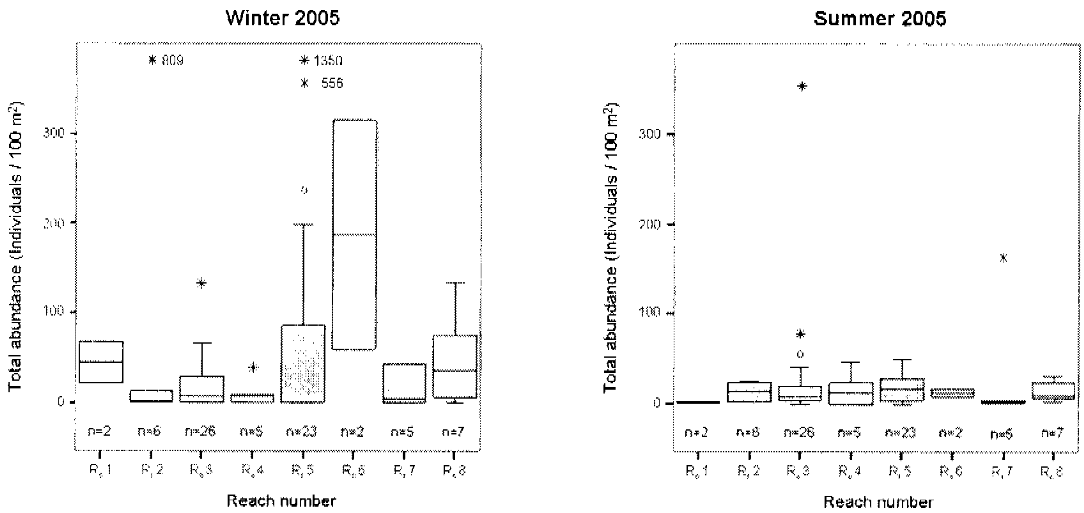

Figure 4: Total fish abundances in winter and summer in the different reaches. Rehabilitated reaches are shown in grey, canalized reaches in white. Circles and stars indicate outliers and extremes, respectively. n = number of sampling stations within each reach.

In winter, only cover and velocity contributed significantly in the overall multiple regression model (p < 0.008, R^2 = 0.740; Table 6): fish densities increased with increasing available cover and with decreasing current velocities.

A significant positive correlation between the total fish abundance and the cover availability was also found in summer (Table 6), but the summer model explained less of the total variance (R^2 = 0.341). No other variables showed any significant relationship.

Table 6: GLM tables for the effects of different abiotic variables on the total fish abundance, separated by season. Significant *p*-values are shown in bold. The variable "Reach number" is nested in the variable "Reach type". The variables "Dist up" and "Dist down" describe the closest distance to a near-natural river reach upstream and downstream, respectively. The distance to the mouth into river Rhine is considered using the variable "Dist mouth".

	Winter		Summer	
Variable	F	*p*	F	*p*
Reach type	0.18	0.677	2.56	0.115
Reach number (Reach type)	1.30	0.273	1.46	0.208
Dom substrate	4.41	0.040	.53	0.470
Flow velocity	7.55	**0.008**	.55	0.459
Dist to mouth	4.57	0.037	3.67	0.060
Dist up	3.83	0.055	5.04	0.028
Dist down	1.02	0.316	.34	0.563
Mean depth	1.26	0.267	.76	0.386
Cover	40.27	**0.000**	9.23	**0.003**
R^2	0.740		0.341	

Only two species, i.e. European chub and spirlin, were sufficiently abundant among the reaches to be studied separately.

Both in winter and summer the abundance of the European chub decreased significantly with increasing distance from the Rhine (Table 7). Similarly, significant differences between reaches were found in both seasons, indicating higher abundances in the reaches located in the lower part of the study area. In winter, the abundance of the chub was significantly positively related with the substrate size and significantly negatively with the flow velocity, while in summer it increased with increasing availability of cover. Furthermore, the winter abundance decreased with increasing distance to the next near-natural reach upstream. Again, the summer model explained less variance than the winter one (R^2 = 0.434 and 0.657, respectively).

A significant positive relationship between the abundance of the spirlin and both the availability of cover and the flow velocity was established in winter (Table 7). In summer, however, the abundance of the sprilin decreased with increasing distance to the next near-natural reach upstream.

Table 7: GLM tables for the effects of different abiotic variables on the abundance of the European chub and the spirlin, separated by season. Significant *p*-values are shown in bold. The variable "Reach number" is nested in the variable "Reach type". The variables "Dist up" and "Dist down" describe the closest distance to a near-natural river reach upstream and downstream, respectively. The distance to the mouth into river Rhine is considered using the variable "Dist mouth".

	Leuciscus cephalus				*Alburnoides bipunctatus*			
	Winter		Summer		Winter		Summer	
Variable	F	*p*	F	*p*	F	*p*	F	*p*
Reach type	0.1	0.758	0.90	0.348	0.12	0.736	1.46	0.231
Reach number (Reach type)	4.52	**0.001**	3.18	**0.009**	0.91	0.497	2.33	0.043
Dom substrate	12.18	**0.001**	2.04	0.159	0.74	0.392	4.31	0.042
Flow velocity	6.44	**0.014**	1.01	0.320	5.98	**0.017**	0.07	0.792
Dist to mouth	18.70	**0.000**	9.71	**0.003**	3.54	0.065	0.20	0.656
Dist up	6.84	**0.011**	4.72	0.034	4.11	0.047	6.17	**0.016**
Dist down	2.89	0.094	0.04	0.850	1.51	0.224	0.02	0.898
Mean depth	3.62	0.062	1.19	0.279	1.27	0.265	0.10	0.759
Cover	2.93	0.092	10.96	**0.002**	27.59	**0.000**	0.16	0.694
R^2	0.657		0.434		0.682		0.318	

Comparison with historical fish data

Six fish and lamprey species are mentioned in the historic sources from the period prior to the systematic correction of the river Thur which are not present in our current samples (Table 5), whereas four recent species are not recorded in the historic source. Three of these are allochthonous species (*Carassius* spp.; *Oncorhynchus mykiss*; *Pseudorasbora parva*). The three-spined stickleback (*Gasterosteus auculeatus*) is a native Swiss species, but according to Steinmann (1936) its historic distribution in the High Rhine did not extend upstream beyond the Basel region, i.e. about 100 km downstream of the mouth of the river Thur. With the exception of P. parva, of which 32 individuals were caught in summer, all of the new species were represented by single individuals.

Additional changes in the assemblage's dominance structure were also observed. Of the four historically most abundant species, only the sprilin was numerously represented in our samplings. Barbel (*Barbus barbus*) and vairone (*Leuciscus souffia*) showed generally smaller abundances with marked seasonal differences. Only a few individuals of the nase (*Chondrostoma nasus)* were found. The barbel reached higher relative abundances in the canalized reaches whereas the vairone was more abundant in the widenings.

The historic sources only provide qualitative information concerning fish abundance. Observations recently made in rivers comparable to the Thur, however, reveal that the majority of abundances observed in this study (Figure 4) are at the low end of the continuum (Zitek and others 2004; A. Peter and E. Schager, unpublished data).
The number and proportion of ecological guilds have also changed as compared with the historical reference (see Table 8 in the appendix). Today's assemblage is highly dominated by individuals of meso-eurythermal species in both reach types, but particularly in the canalized reaches. Oligo-stenothermal species are numerically underrepresented. Piscivorous species such as *Esox lucius* are virtually lacking today whereas the proportion of omnivorous species is higher in both reach types. Species that are tolerant to various anthropogenic impacts are generally more abundant than they were around 1850, particularly in the rehabilitated reaches.
The results above reveal that the fish assemblage in both canalized and rehabilitated reaches is still quite far removed from near-natural reference conditions.

Discussion

Habitat set

Habitat diversity

Diversity measures for canalized and rehabilitated reaches overlap as there is a considerable variation within both reach types. R_r 2, the oldest and shortest widening, is comparable with the canalized reaches, whereas the two other widenings, R_r 5 and R_r 7, display higher habitat diversities. This is not only due to their elevated number of habitat types, but also to their more balanced distribution as demonstrated by the higher evenness. The canalized reaches and R_r 2 are dominated by medium flow velocities, whereas in R_r 5 and R_r 7 the relative amount of both slow and fast flowing habitats is higher. Shallow water areas are generally underrepresented in canalized reaches and in R_r 2, whereas they compose a higher proportion in the rehabilitated reaches R_r 5 and R_r 7. These results coincide with the conclusion of Grift and others (2001) that shallow slow-flowing areas offering suitable spawning and rearing conditions for rheophilic fish are severely degraded or even absent in regulated rivers.

Habitat recovery

In natural systems, habitats are a function of various geomorphological processes operating on different temporal and spatial scales (Frissel and others 1986). Thus, habitat recovery, i.e. the development of characteristic habitats after rehabilitation, requires space and may take several years. In the current case, no relationship was observed between habitat diversity and time span since rehabilitation: R_r 2 and R_r 7 are about the same age, but their diversities differ clearly, whereas R_r 5 and R_r 7 differ in age but their diversities are comparable. We assume that the habitat set in the youngest widening R_r 5 (completed in 2002) will further diversify with every major flood event, in particular at the natural river banks (erosion, establishment of riparian vegetation), whereas in the two older widenings the habitat potential was probably achieved during the 13 - 14 years of development.

Size of widenings

There is evidence that the spatial dimension of the widening influences habitat diversity. With a length of about 0.3 km, the widening with the lowest diversity, R_r 2, is considerably shorter than the more diverse reaches R_r 5 and R_r 7 (1.9 km and 0.9 km). However, more replicates of widenings of different lengths would be required to test this hypothesis. Based on laboratory experiments, Hunzinger (1998) suggests a minimal widening length of 420 m in order to initiate channel braiding at the river Thur. This is consistent with the recent strategy of the local river managers to build longer and larger widenings. R_r 2 is shorter than the minimal requirements for braiding, and this leads to the development of alternating gravel banks instead. This morphology is further stabilized by the bed load deficit in this part of the Thur (Schälchli 2005) and the location of R_r 2 in a slight bend.

Reference conditions

Expressed by the shoreline length, the rehabilitated reaches attained a generally higher degree of naturalness than the canalized reaches. Despite this improved lateral connectivity, rehabilitated reaches are still far from achieving near-natural conditions, as the low percentages from the reference values demonstrate.

Shorter current shorelines are due to the virtual lack of multiple channels, groundwater channels and side arms and to the low number of backwaters or isolated floodplain water bodies. As discussed above, the banks in the latest widening, R_r 5, will probably diversify further and possibly lead to a higher naturalness value in the future.

In general, shoreline length appears to be a good indicator for overall habitat diversity, H_S. However, the extremely low percentages of R_c 1 and R_r 2 within their reach types must be treated with care. Due to their short length (300 m), the reference reaches were not representative of the historically quite significantly braided reach. Compared with a larger-scale reference (historical reaches 1 to 3 combined), ratios of 1 % and 15 % result for R_c 1 and R_r 2, respectively, confirming the general trend of a slightly higher naturalness in rehabilitated reaches. The relatively small reference value in the lowest reach R_c 8, however, is not biased by reach length but due to the reach's position at the transition from the braided to the meandering channel type which is naturally characterised by shorter shorelines.

Fish assemblages

Total fish abundances and species numbers in all canalized and rehabilitated reaches in this study were comparable, even in R_r 5 and R_r 7 which displayed a more diverse habitat supply. Assemblage structure in all reaches was assessed as being far from reference conditions.

Critical concerns

a) Absence of species: Most of the historical species that are missing in our catches were of low historical abundance (Table 5). As the occurrence of rare species varies greatly over time (Jackson and others 2001), their absence should not be weighted too negatively. Furthermore, single individuals of grayling (A. Peter, unpublished data), brook lamprey, perch and sculpin were observed recently in the river Thur (Schager and Peter 2005). Although absent in our extensive sampling, these species are still present in the system, representing an important recolonization source for a future assemblage recovery (Niemi and others 1990).

The two anadromous species, Atlantic salmon and European river lamprey, however, became extinct on a nationwide scale. Re-colonisation may only take place by large-scale, transboundary rehabilitation measures (Nienhuis and others 2002b). Despite the absence of several taxa, the species richness in the Thur is still surprisingly high for a river that has been almost completely canalized for many decades.

b) Reproductive success: By excluding the young of the year fish (YOY), our study did not cover the species' complete life cycles. Due to higher proportion of suitable spawning and rearing grounds, higher YOY densities might be expected in the two longer widenings, i.e. R_r 5 and R_r 7. Corresponding findings were already reported for small scale rehabilitation measures in the river Thur (Rey and Ortlepp 1999). However, we did not detect such differences in the older age classes (> YOY). The observed dominance of young individuals could be attributed to a reduced catchability of older fish, particularly in summer. During our extensive field campaigns, partly undertaken by zodiac, however, we observed only single schools of larger-sized rheophilic fish, indicating a reduced density or a temporally limited occurrence of adult individuals in the river Thur.

Low abundances

Observed total abundances were generally below the values of comparable near-natural reference rivers, especially in summer. Given that macroinvertebrates are available with high abundances and a rich diversity (Baur 2002; Lubini 1994; Uhlmann 2001) which even appear to be slightly elevated in longer widenings (Limnex AG 2006), food resources for the mainly insectivorous species in the historical data set do not seem to be limited. Although organic loads may be pronounced at low flow (Dreyer 2003), water quality conditions are generally not detrimental to aquatic life in the river Thur. However, a strong positive relationship was observed between the total abundance of fish and the availability of cover in both seasons. Due to the riprap bank reinforcement, cover availability is slightly elevated in the canalized reaches as compared with the three widenings. Because of a reduced input at the mostly fixed shores and removal at the weirs, woody debris is largely lacking as a naturally structuring element (Bisson and others 1987) in the Thur system.

The relative importance of cover in relation to the total fish abundance varied seasonally. The highly clustered distribution of fish in winter can be explained satisfactorily by the abiotic variables we used. Most species–including those classified as moderately structure-dependent such as the nase or the barbel–retreat to well structured, slow flowing or standing water habitats which provide shelter from hydraulic stress (Jungwirth and others 1995) and other adverse physicochemical conditions (Cunjak 1996). In particular, structure-rich backwaters and isolated pools of medium to high depth constitute winter hotspots (Cunjak 1996; Lusk and others 2001) where we found total fish densities > 500 individuals / 100 m^2. Furthermore, backwaters may serve as flood refugia throughout the year (Lusk and others 2001) increasing the assemblage's resilience to flood events (Sedell and others 1990; Townsend and others 1997). As these parapotamal habitats suffered considerably from canalization works (Aarts and others 2004), we only found them in the three widened reaches, underlying the local and maybe even regional significance of the rehabilitated river reaches (Cunjak 1996; Lusk and others 2001).

As indicated by the low level of explained variance, however, our model lacks important factors that influence the abundance of fish in summer. Due to their higher activity levels, it is possible that the summer distribution of fish was far more significantly driven by biotic interactions, such as intra- and inter-specific competition, than by abiotic factors. However, the abundances observed at most stations were so low as to possibly render competition unimportant (Jackson and others 2001). Additional unconsidered abiotic variables, such as thermal heterogeneity, could therefore contribute to the observed patterns and to the generally lower abundances in summer. In contrast to natural braided systems where lateral temperature differences of up to 15 °C may be observed in summer (Arscott and others 2001), the river Thur displays a relatively homogeneous temperature pattern (Frey and others 2003). Considerable temperature changes only occur at the few confluences of groundwater-fed drainage canals or in shallow, slow-flowing waters along gravel banks (Frey and others 2003). Summer temperatures regularly exceed 21 °C, an effect that has even increased over the past two decades (BUWAL and others 2004).

Based on the historical maps we assume that a higher and larger-scale hyporheic exchange existed prior to canalization creating thermal refugia in both cold (Craig and Poulin 1975; Cunjak and Power 1986) and hot seasons (Gibson 1966; Nielson and others 1994). This assumption is strongly supported by the reduced relative (summer) abundance of oligo-stenothermal species, such as the vairone, the brown trout and the minnow, observable today and by the absence of the formerly abundant sculpin. Thus, future rehabilitation measures should address the increase in the thermal heterogeneity, in particular.

Changes in dominance structure and guild composition

Considerable changes in the assemblage's dominance structure and hence in its guild composition were documented. The observed increase in generalist and tolerant species, which is mainly due to the high relative abundance of the omnivorous tolerant European chub, and the lower representation of specialists such as piscivorous or insectivorous taxa, is very familiar from numerous rivers suffering from various human impacts (Paller and others 2000; Shields and others 1997). Apart from the thermal changes discussed above, migration barriers and degraded habitat conditions (Gerster 1998) in the impounded High Rhine may also be responsible for the under-representation of formerly abundant migratory species such as the barbel and the nase which exists today. In 1995/96 only 296 individuals of nase were counted in a one-year-survey carried out at the 12 fish passes between Basel and the mouth of the river Thur (ca. 100 km; Gerster 1998); in contrast, the number of barbels recorded (39,731 individuals) was considerably higher.

Conclusions

This study demonstrated that aquatic habitat diversity may be improved by local river widening provided that a minimal river-specific dimensioning is exceeded. Structure-oriented fish species such as brown trout or grayling could respond rapidly to such improvements, however their density in the river Thur is of no importance. The response of the river Thur fish assemblage was weak indicating that there could be ongoing deficits such as reduced thermal heterogeneity, or the failure of the faunal recovery process to be completed.

Furthermore, the habitat condition of the entire river Thur must be considered: 65 % of the lower 90 km of the river still display degraded morphological conditions and/or are impaired by fragmentation or residual flow. Therefore, we assume that a strong reaction of the total fish fauna could only be achieved by large-scale habitat rehabilitation. Nienhuis (2002a) stated that it could be beneficial if the predicted rehabilitation outcome fails to materialize as further "research [is] needed to understand the mechanisms and learn the constraints in the ecosystem, at the benefit of future predictions by both manager and researcher".
Reference comparison was proved to be an important prerequisite for the evaluation of the degree of naturalness, i.e. the success of rehabilitation.

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Appendix

Table 8: Guilds of fish and lamprey species in the river Thur and their change in occurrence from 1890 to present. Guild assignment according to Schmutz and others (2000) and Woolsey and others (2005). Historical occurrences derived from Wehrli (1892; Table 5). ↑ = increase; ↓ = decrease; - = no change.

	Historical occurrence	Temperature	Habitat	Reprod.	Feeding	Migration	Tolerance	Structure	Longevity	Change in occurrence	
										can	reh
Cottus gobio	high	OLIGO	RHEO	SPEL	BENT-INS	SHO	INT	DEP	L-SHO	↓	↓
Leuciscus souffia souffia	very high	OLIGO	RHEO	LITH	BENT-INS	SHO	INT	M-DEP	L-MED	↓	↓
Oncorhynchus mykiss	non-native	OLIGO	RHEO	LITH	BENT-INS	MED	INT	M-DEP	L-MED	↑	-
Phoxinus phoxinus	high	OLIGO	RHEO	LITH	BENT-INS	SHO	TOL	M-DEP	L-SH	-	-
Salmo trutta fario	medium	OLIGO	RHEO	LITH	BENT-INS	SHO	INT	DEP.	L-MED	↓	↓
Salmo salar	low (seasonal)	OLIGO	RHEO	LITH	BENT-INS	LONG	INT	DEP	L-MED	↓	↓
Thymallus thymallus	low	OLIGO	RHEO	LITH	BENT-INS	MED	INT	M-DEP	L-MED	↓	↓
Alburnoides bipunctatus	very high	MESO	RHEO	LITH	BENT-INS	SHO	INT	M-DEP	L-SH	-	-
Alburnus alburnus	low	MESO	EURY	PHYT	OMNI	SHO	TOL	IND	L-SH	-	-
Anguilla anguilla	medium	MESO	EURY	PEL	OMNI	LONG	TOL	DEP	L-MED	-	-
Barbatula barbatula	high	MESO	RHEO	LITH	BENT-INS	SHO	TOL	M-DEP	L-MED	↓	↓
Barbus barbus	very high	MESO	RHEO	LITH	BENT-INS	MED	INT	M-DEP	L-LO	↓	↓
Carassius spp.	non-native	MESO	LIM	PHYT	OMNI	SHO	TOL	M-DEP	L-MED	↑	-
Chondrostoma nasus	very high	MESO	RHEO	LITH	HERB	MED	INT	M-DEP	L-MED	↓	↓
Esox lucius	high	MESO	EURY	PHYT	PISC	SHO	TOL	DEP	L-LO	↓	↓
Gasterosteus auculeatus	unmentioned	MESO	EURY	PHYT	OMNI	SHO	TOL	IND	L-SH	↑	↑
Gobio gobio	high	MESO	RHEO	PSAM	BENT-INS	SHO	TOL	M-DEP	L-SH	↓	↓
Lampetra fluviatilis	low (seasonal)	MESO	RHEO	LITH	PISC	LONG	INT	IND	L-MED	↓	↓
Lampetra planeri	medium	MESO	RHEO	LITH	DETR	MED	INT	IND	L-MED	↓	↓
Leuciscus cephalus	high	MESO	RHEO	LITH	OMNI	MED	TOL	DEP	L-MED	↑	↑
Leuciscus leuciscus	low	MESO	RHEO	LITH	OMNI	SHO	INT	M-DEP	L-MED	-	-
Perca fluviatilis	low	MESO	EURY	PHYT	BENT-INS	SHO	TOL	IND	L-MED	-	-
Pseudorasbora parva	non-native	MESO	LIM	POLY	OMNI	SHO	TOL	IND	L-SH	↑	↑
Rutilus rutilus	low	MESO	EURY	PHYT	OMNI	SHO	TOL	IND	L-MED	-	-
Scardinius erythrophthalmus	low	MESO	LIM	PHYT	OMNI	SHO	TOL	M-DEP	L-MED	-	-
Tinca tinca	restricted to lakes	MESO	LIM	PHYT	OMNI	SHO	TOL	DEP	L-LO	-	-

OLIGO = oligo-stenotherme; MESO = meso-eurytherme; RHEO = rheophilic; EURY = eurytopic; LIM = limnophilic; SPEL = speleophilic; LITH = lithophilic; PHYT = phytophilic; PSAM = psammophilic; PEL = pelagophilic; POLY = polyphilic; BENT-INS = benthic-insectivorous; OMNI = omnivorous; HERB = herbivorous; PISC = piscivorous; DETR = detritivorous; SHO = short-distance; MED = medium-distance; LONG = long-distance; INT = intolerant; TOL = tolerant; DEP = structure-dependent; M-DEP = moderately structure dependent; IND = structure-independent; L-SH = short-lived; L-MED = intermediate livetime; L-LO = long-lived.

CHAPTER 5

Recovery rates in riverine fish assemblages following physical habitat rehabilitation: A review of selected case studies

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Abstract

In the present review, we analysed 35 selected case studies dealing with the recovery of riverine fish assemblages following active rehabilitation of physical habitat and lateral connectivity. In particular, we were interested in how the choice of different types of indicators and reference conditions might influence the investigators' conclusions on rehabilitation outcome and recovery time. In most studies, fish response was measured at the population level. Structural and compositional indicators dominated while functional indicators were highly underrepresented. Both near-natural and degraded reference conditions were used to detect rehabilitation-induced changes. In several studies, multiple types of references were applied, leading to sometimes inconsistent evaluation of rehabilitation outcome. In each study, at least one recovery endpoint was achieved. Inconsistencies in data availability and temporal resolution as well as a great variability within different indicator types led to a highly heterogeneous data set. Therefore, a general statement concerning the effect of using different indicator types on the project evaluation was hampered.

We conclude with implications and guidelines for future river management, propose monitoring activities, and identify research needs.

Keywords: River rehabilitation, scale, indicator, endpoint, recolonisation

1. Introduction

Running waters are strongly affected by human impacts (Benke, 1990; Nilsson *et al.*, 2005b), but are considered as highly resilient to various disturbances (Ward *et al.*, 2001). Therefore, rivers and streams seem to offer a great potential for rehabilitation projects (Cairns, 1990), i.e. for a re-initiation of the natural recovery process after sufficient degradation. In their comprehensive review, Roni *et al.* (2005) showed that rehabilitation of riverine habitats is generally of varying biological effectiveness. In some projects, the biology completely recovered. In others it did not. This observation is in contrast to the results in morphology where more consistent, generally positive responses were observed (Roni *et al.*, 2005). For identical measures the biological outcome varied. The heterogeneous biological results could therefore not be explained by the measures applied (Roni *et al.*, 2005).

Comparisons between different rehabilitation projects are crucial despite, or even because of, such inconsistencies. Driving factors and sensitive elements of the recovery process as well as conceptual deficiencies can only be identified by comparing various experiences. This information is badly needed to further improve predictive capacity in river rehabilitation, i.e. to reconcile practical observations with theoretical models (Yount and Niemi, 1990). Rehabilitation projects afford an ideal opportunity to test the understanding of ecological patterns and processes in riverine systems (Bradshaw, 1996). This is of particular importance given the growing number of river rehabilitation projects worldwide (Bernhardt *et al.*, 2005; Nakamura *et al.*, 2006; Nienhuis and Leuven, 2001) and the urgent need to rehabilitate degraded river courses (Benke, 1990; Nilsson *et al.*, 2005b; Peter *et al.*, 2005; Tockner and Stanford, 2002).

Various authors identified three areas that are not adequately considered in most river rehabilitation projects (Bash and Ryan, 2002; Kondolf, 1995; Kondolf and Downs, 1996; Roni *et al.*, 2005). 1) A comprehensive evaluation of the actual degradation before project start. 2) The influence of watershed processes on local projects. 3) A thoroughly planned, well-funded monitoring of the outcome. As a result, many important aspects of system recovery remain poorly understood (O'Neill, 1999). This is particularly the case for factors influencing the recovery process and the rehabilitation outcome (*Lepori et al., 2005*).

In this review of selected case studies we specifically addressed the temporal process of recovery in riverine fish assemblages following rehabilitation of the physical habitat. First, we were interested in the general rehabilitation outcome in the reviewed studies, i.e. the investigators' evaluation of rehabilitation effectiveness. Was a recovery of the measured fish indicators observed, i.e. were the predefined endpoints reached? As information on system development is always dependent on the monitoring design applied, we shortly discussed the study designs that were used. Second, we tried to identify the time span from rehabilitation to endpoint achievement, i.e. the recovery rate. Third, we asked to what extent the observed results might be influenced by the indicators, reference types and endpoints used. Finally, we collated those factors that investigators reported to account for the observed rehabilitation outcome. Based on our findings we derived implications for future river management, project planning and monitoring activities and identified research needs.

2. Disturbance and recovery: Some definitions

Many different terms are tied to the process of recovery. As they are used in distinct ways, we shortly discuss important definitions and clarify their use in the present paper.

In running waters, those events may be called a *disturbance* that are relatively discrete in time and that exceed the predictable range of frequency, severity or intensity (Resh *et al.*, 1988). Disturbances disrupt the structure of the ecosystem, community, or population and change resources, the availability of substratum or the physical environment (Pickett and White, 1985).

Two types of disturbances may be distinguished (Bender *et al.*, 1984; Yount and Niemi, 1990) which differ in their duration and their effect on the biota. *Pulse* disturbances are relatively short-term events, such as inputs of non-persistent pollutants (Milner, 1994). Such events lead to an instantaneous alteration of system attributes, such as the density of certain species.

In contrast, *press* disturbances persist over the long-term, i.e. longer than the life time of the longest-lived species in a community, and represent a sustained alteration of system attributes. Direct human impacts on river morphology, such as canalisation, logging or mining activities can be classed as press disturbances (Niemi *et al.*, 1990).

If the disturbance is no longer active, recovery can occur. Generally, *recovery* may be defined as the return toward undisturbed conditions (Yount and Niemi, 1990), that is, toward nominal behaviour (Gerritsen and Patten, 1985). Recovery is considered to be completed when a predefined endpoint is achieved (Niemi *et al.*, 1990). Such an *endpoint* may be represented by a value or range of any biological, physical or chemical variable of interest.

The pace at which the recovery takes place (*recovery rate*) depends on the characteristics of the disturbance, such as its duration or the severity of habitat alteration (Detenbeck, 1992). After sufficient degradation a system's capacity to naturally recover from disturbance (*resilience*) may be lost (Kauffman *et al.*, 1997). In this case, the recovery process must be re-initiated by active human manipulation (Gore, 1985; Kauffman *et al.*, 1997), i.e. by restoration measures.

Restoration is generally defined as the process of bringing a system back to its pristine, intact state (Bradshaw, 1997; Roni *et al.*, 2005) or historic trajectory (SER, 2005). Due to infrastructural and financial constraints, this is often difficult or even impossible to achieve in practice (Stanford *et al.*, 1996). *Rehabilitation* therefore describes the approach of improving ecosystem composition, structure and function to a near-natural, but not pristine state (Bradshaw, 1997). As in restoration, this has to result in a system with integrity (Angermeier, 1997), i.e. a dynamic, ecologically healthy river (Palmer *et al.*, 2005). We use the term 'rehabilitation' throughout this article to refer to the different approaches used in the case studies.

To assess rehabilitation outcome, reference information is required which characterises the biotic and abiotic conditions that are aspired by rehabilitation ('Leitbild' or guiding image; Jungwirth *et al.*, 2002). Reference information may be derived from relatively undisturbed conditions (hereafter referred to as 'near-natural') on the one hand, and degraded conditions on the other hand (Table 1).

Table 1: Sources to derive reference information from (slightly modified from Palmer *et al.*, 2005).

1. Near-natural reference conditions:

   Mirror the relatively undisturbed conditions for a given site. Such information might be gained from:

   a. historical, pre-disturbance data of the specific locality consulting aerial photographs, documentary sources, topographic maps
   b. current undisturbed sites that are comparable regarding geology, climate, hydrology and zoogeography
   c. theoretical information derived from conceptual or empirical models or classification systems.

2. Degraded reference conditions:

   Describe the conditions at a current impaired, but untreated site that has similar system characteristics (e.g. geomorphology, hydrology, zoogeography) to the rehabilitated site. This type of reference might be used when a reference condition "to move away from" is needed. The term 'control site' is used synonymously.

3. Selection of case studies

A variety of different approaches and techniques is in use to rehabilitate degraded riverine habitats (Table 2). These measures address specific ecosystem characteristics, such as the longitudinal connectivity or the flow regime. Depending on the project objectives, the selected methods and measurements for monitoring differ. In order to compare various projects, a focus on a specific rehabilitation objective therefore seems appropriate. In this review, we concentrate on schemes which are intended to actively improve the structural diversity and lateral connectivity in anthropogenically impaired lotic systems (Table 2). Such structural measures are of worldwide importance given the high number of rivers and streams affected by morphological impairments such as canalisation or stream clearing.

Case studies were identified by a systematic search of relevant books and the peer-reviewed literature utilising the ISI Web of science database. We restricted our attention to case studies that (i) address the recovery of the fish fauna and (ii) provide a minimum of information on project design and recovery process (i.e. indicators used, project duration, observed changes). As our focus lay on rehabilitation of anthropogenically disturbed systems, we excluded the large amount of studies which describe river restructuring without explicitly indicating a previous anthropogenic impact (e.g. Avery, 1996; Binns, 1994; Hunt, 1976; Solazzi *et al.*, 2000).

Table 2: Rehabilitation measures to improve degraded riverine habitats (modified from Woolsey *et al.*, 2005 and Roni *et al.*, 2005). For the techniques discussed, the number of reviewed case studies and the percentage of the total number of reviewed articles (in parentheses) are given. In several projects, multiple techniques were applied.

Category and techniques used	No. of reviewed articles (%)
Flow regime	
Reestablishment of natural flow regime, increase of residual flow, attenuation of hydropeaking	-
Structural diversity and lateral connectivity	
Reconnection of floodplain water bodies, creation of new floodplain habitats, local river widening, re-meandering	12 (34)
Placement of boulder or wood structures, spawning gravel and brushwood	29 (83)
Reestablishment of riparian vegetation, i.e. planting of riparian trees or abandonment of grazing	1 (3)
Reopening of piped streams	-
Longitudinal connectivity	
Removal of migration barriers (e.g. dams, weirs or local culverts at road crossings), construction of fish passes and bypass systems	-
Bed load regime	
Reestablishment of natural bed load regime	-

A total of 35 articles was found, published between 1986 and 2006 (Figure 1). This constitutes only a small part of the worldwide literature on the recovery of fish assemblages following habitat rehabilitation. Unfortunately, most of the relevant information is not easily accessible as it is published as grey literature only (Bernhardt *et al.*, 2005) and is often exclusively available in the local language. By considering previous review articles in the discussion, some information from reports and compendia is nevertheless incorporated.

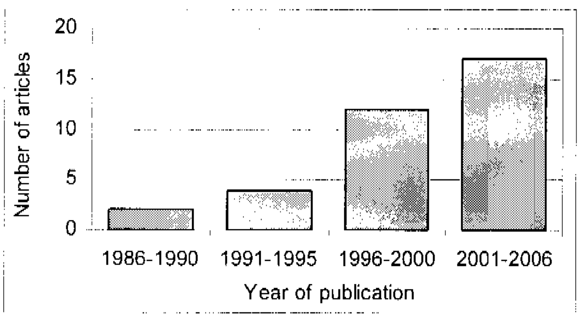

Figure 1: Reviewed case studies according to their year of publication.

We believe the present collection to be a representative extract of the currently published peer-reviewed literature on the selected rehabilitation techniques (Table 2). Accordingly, the placement of instream structures as one of the most common and widespread rehabilitation techniques in use (Roni *et al.*, 2005) is represented in the majority of the articles (83 %;Table 2). For other techniques, less experience or information is available. The reopening of piped streams, for instance, is a measure that is quite frequently used in European countries. There, small streams have been piped over large courses to reclaim land for agriculture, e.g. 17 % of the total Swiss river network (Peter, 2006). To our knowledge, no peer-reviewed case studies have been available referring to this subject so far.

The rivers systems described in the case studies are located in North-America (20 articles; 57 %), Europe (14 articles; 40 %) and in Australia (1 article, 3 %). In most case studies, fourth- to fifth-order rivers of medium gradient were investigated (Table 3). In the studied systems, canalisation was the anthropogenic impact which was predominantly responsible for the degradation of the physical habitat (Table 3).

Table 3: Characteristics of the systems and projects presented in the case studies.

1. Characteristics of study rivers and streams [1)]

Parameter	Median	Range	No. of articles (%)
Stream order	4	2 - 9	16 (46) [2)]
Average annual flow [$m^3 \cdot s^{-1}$]	30	0.4 – 1'900	10 (29) [2)]
Elevation [m.a.s.l]	325	18 – 2'675	7 (20) [2)]
Average gradient [‰]	2.3	0.4 – 29.23	17 (49) [2)]
Catchment size [km^2]	71.2	9.3 – 102'000	20 (57) [2)]
Salmonid streams [3)]			17 (49)
Length of rehabilitated sites [m]	1'000	30.5 – 4'200	22 (63) [2)]

2. Anthropogenic impacts responsible for habitat degradation [4)]

Impact	Affected relevant processes (exp.)	No. of articles (%)
Canalisation	Channel and floodplain interactions	19 (54)
Stream clearing for timber floating	Channel and floodplain interactions	6 (17)
Logging activities	Wood delivery	11 (31)
Others (e.g. reservoir operation, thermal pollution)	Sediment supply, riparian functions	14 (40)

[1)] Mean values were used for articles where multiple sites or systems were sampled.
[2)] Limited data availability due to lack of information in the case studies.
[3)] Assemblage dominated by salmonid species.
[4)] In several systems, multiple disturbances were active.

4. Comparison of monitoring schemes

To simplify orientation, citations addressing reviewed case studies are written in italics.

4.1 Indicators

A total of 42 indicators for monitoring recovery in fish assemblages was evaluated in the case studies (Table 4, see Table 8 in the appendix for the complete list). These indicators address different levels of the ecological hierarchy (Angermeier, 1997): population, guild and community (Table 4). At each hierarchical level, compositional, structural and functional indicators can be distinguished (ecosystem attributes; Dale and Beyeler, 2001; Noss, 1990) leading to a total of 9 indicator types (Table 4). Compositional indicators describe the identity and variety of biotic elements (e.g. species diversity, total abundance and biomass) whereas their physical organisation or pattern, such as age distribution is characterised by structural indicators (Noss, 1990). Finally, functional indicators are direct measures of ecological and evolutionary processes (Noss, 1990). Examples of functional fish indicators are: growth rate, movement and production.

Table 4: Characteristics of the indicators used in the case studies. Nine different types of indicators can be distinguished depending on the hierarchical level and the ecosystem attribute they address.

<------------------------ Ecosystem attribute ------------------------>

<- Ecological hierarchy ->	Composition	Structure	Function	Total
Community	13 indicators (19 articles, 54 %)	9 indicators (9 articles, 26 %)	0 indicator	22 indicators (21 articles, 60 %)
Guild	0 indicator	2 indicators (2 articles, 6 %)	0 indicator	2 indicators (2 articles, 6 %)
Population	2 indicators (5 articles, 14 %)	12 indicators (24 articles, 69 %)	4 indicators (5 articles, 14 %)	18 indicators (27 articles, 77 %)
Total	15 indicators (22 articles, 63 %)	23 indicators (28 articles, 80 %)	4 indicators (5 articles, 14 %)	42 indicators

Structural and compositional indicators dominate with respect to both the variety of indicators and their use in the case studies (Table 4 and Table 8). In most cases (27 articles), fish response was measured on the population level, whereas guild indicators were surveyed in 2 cases. However, information on guild assignment was utilised much more frequently. As it was usually expressed as relative abundance, we treated it as a community and not as a guild indicator (Table 8).

Individual indicators were measured in different measurement units, such as the indicator species richness which was, among others, expressed as the absolute number of species at a given site or their presence-absence pattern, the number of species per metre of river course, the number of species per 100 individuals or the number of species for a common number of individuals (calculated e.g. by rarefaction curves).

Functional indicators are highly underrepresented in our sample although the reestablishment of disrupted ecosystem processes and functions is widely postulated in the reviewed case studies and elsewhere in the literature (Angermeier, 1997; Palmer *et al.*, 2005; Roni *et al.*, 2002; Ward *et al.*, 2001). Several factors might be responsible for this. The use of structural and compositional indicators has a long tradition in fisheries science and practice, whereas the significance of complex riverine processes on ecosystem composition and structure has only recently been appreciated and more intensively studied (Ward *et al.*, 2001). This results in a still lower availability and recognition of functional indicators (see *Gowan and Fausch, 1996* and Trexler, 1995 for examples of functional fish indicators for assessing rehabilitation outcome). The measurement of functional indicators is often expensive (Angermeier and Karr, 1994) because it requires intensive sampling and analysis. However, existing techniques are continually enhanced and newer approaches, such as stable isotope or genetic analyses, offer a great potential for studying functional relationships (Bilby *et al.*, 1996; Depledge, 1999; Fry, 2002).

In the literature, several benefits from functional indicators are accentuated. Kelly and Harwell (1990) emphasise the utility of process-oriented indicators especially for the early stages of recovery from (ecotoxic) disturbances as they directly indicate the presence of ongoing or new stresses ("early warning indicators"). In an experimental macroinvertebrate study, functional indicators exhibited a lower spatio-temporal variance between replicate treatments and a higher responsiveness to changes in physical habitat heterogeneity than structural and compositional indicators (Brooks *et* *al.*, 2002). Furthermore, functional indicators enabled the identification of the causes of the variability in structural and compositional indicators, thereby preventing false conclusions on ecosystem health (Bunn and Davies, 2000).

In 14 studies, indicators from a single indicator type were measured, i.e. the studies addressed a single ecosystem attribute within a certain hierarchical level (Table 4). In the literature, however, the use of indicators from several hierarchical levels is recommended in order to fully account for the system's response (Angermeier, 1997).

4.2 Design and reference information

Case studies were assigned to one of two monitoring designs depending on the temporal pattern of data collection (Table 5). In a before-after design (BA), rehabilitated sites are sampled both before and after project implementation while the post-treatment design (PT) represents a retrospective approach where no pre-project data are included (Roni, 2005). In our sample, there is an almost equal representation of both monitoring types (Table 5).

Table 5: Monitoring design and reference information used in the reviewed case studies. A description of the two reference types is given in Table 1.

Reference type used		Before–after design (BA)	Post-treatment design (PT)	Total
Degraded conditions	Near-natural conditions	No. of articles (%)	No. of articles (%)	No. of articles (%)
yes	no	10 (29)	8 (23)	18 (51)
no	yes	2 (6)	5 (14)	7 (20)
yes	yes	5 (14)	4 (11)	9 (26)
no	no	1 (3)	0 (0)	1 (3)
	Total:	18 (51)	17 (49)	35 (100)

With one exception, reference information was used in all case studies (Table 5). In 27 articles (77 %) degraded reference sites were sampled, e.g. by means of a BACI sampling (Stewart-Oaten *et al.*, 1986). Near-natural reference conditions were considered less frequently (16 articles or 46 %). In most of these cases (9 articles), current near-natural sites were sampled. In 6 case studies, theoretical references, such as conceptual models, were used whereas historic information was drawn on in 2 cases.

In 13 studies (37 %), rehabilitation was not spatially replicated, i.e. a single rehabilitated site was sampled. Eighteen studies (51 %) investigated ≥ 2 spatially separated rehabilitation sites, and for 4 studies (11 %) it was not apparent whether rehabilitated sites were replicated. From the 27 studies that sampled controls, 10 utilised multiple sites while 13 and 4 studies applied no or an unclear number of replicates, respectively. Nine studies considered current near-natural sites; four of them had no replicates while in five ≥ 2 sites were sampled.

BA-studies were generally of medium post-project duration (35 months, Table 6), whereas long-term studies of > 5 years post-project duration were the exception (4 cases). On average, 2 pre- and 6 post-project samplings were carried out. In 6 and 2 BA-studies, sites were only sampled once before and after project implementation, respectively.

PT-studies were usually of shorter duration, i.e. 15 months on average, and normally started about 1.5 years after project implementation. Five PT-studies comprised a single survey of the sites, i.e. temporal replication was not carried out.

Table 6: Monitoring duration, intensity and time of sampling in the reviewed case studies. Mean values were used for articles where multiple sites were sampled over different time scales.

	Design type [1]	Min	Max	Median	No. of articles (%)
Pre-project monitoring					
Duration [mo] [2]	BA	6	96	24	12 (100)
Number temporal replicates	BA	1	11	2	17 (94) [3]
Post-project monitoring					
Duration [mo] [2]	BA	6	94	35	16 (100)
	PT	1	53	15	12 (100)
Number temporal replicates	BA	1	13	6	17 (94) [3]
	PT	1	60	6	14 (82) [3]
First sampling [mo]	BA	0.5	12	8.75	18 (100)
	PT	1	108	19.5	16 (94) [3]

[1] BA = before-after design, PT = post-treatment design.
[2] Only for studies with ≥ 2 temporal replicates in the pre- and post-project monitoring, respectively.
[3] Limited data availability due to lack of information in the case studies of the particular design type.

The case studies used different reference types which vary in their quality and meaning. The simultaneous sampling of current near-natural or degraded reference sites allows for the quantification of natural variation and change (Chapman, 1999). These must be distinguished from the effects of rehabilitation (Roni, 2005).

Control sites are important for recognizing rehabilitation-independent factors which might have generally influenced the degraded river reaches (Chapman, 1999). The consideration of any type of near-natural reference (Table 1, 1a -c) enables the evaluation of the degree of naturalness or degradation of both the rehabilitated (pre- and post-project monitoring) and the control sites. If possible, the use of both reference and control sites is recommended (Chapman, 1999).

To reliably identify any fluctuations in time, appropriate temporal replication is needed (Underwood, 1992), both in the pre- and post-project monitoring. This requires that multiple samplings be made, an assumption that is not met in all case studies. To fully account for seasonal differences, Roni *et al.* (2002) recommended to consider particularly the critical life stages, e.g. winter and spring for juvenile salmonids. Additionally, temporal replication implies that fish populations need to be followed over a sufficiently long time span. Reeves *et al.* (1991) suggested that monitoring should be conducted over a minimum of two generations of the fish species of interest, i.e. 6 to 10 years for certain salmonid species. Compared to these figures, most of the reviewed studies were of relatively short duration, particularly in terms of the pre-project monitoring.

Spatial replicates of reference and control sites –and ideally also of rehabilitated sections - enable to account for inherent, location-specific differences between sites (Underwood, 1992). This is especially true for PT-studies where the assumed pre-project similarity of treatment and control sites cannot be verified in retrospect (Roni, 2005). An example for a temporally and spatially replicated case study is given by *Gowan and Fausch (1996)* who investigated pairs of rehabilitated and untreated, degraded reaches over 8 years in 6 Colorado streams (2 years before, 6 years after project implementation). They identified regional, project-independent factors leading to a population decline over all reach types in the second half of the post-project monitoring.

4.3 Endpoints

In order to identify whether recovery has been achieved or not, endpoints need to be defined for the indicators used. Here, we were interested in the investigators' procedure for establishing standards. In most case studies, endpoints were reference-specific, i.e. only few indicators were evaluated independently of reference information.

Depending on the mode of reference comparison, two general categories of endpoints can be distinguished, statistic-based and descriptive endpoints.

Table 7: Endpoints used in the reviewed case studies to identify recovery achievement

1. Statistics-based endpoints:

   Indicator measurements from the different types of sites were compared by means of statistical tests. This requires the availability of numerical data for both rehabilitation and reference sites. Two types of statistics-based endpoints were used:

   a. The endpoint was derived from the situation when *no statistical significant difference* was observed between the indicator values of the different types of sites: This endpoint was used to identify the achievement of near-natural reference conditions. Due to the requirement of numerical data, its application was mainly restricted to studies that comprised current near-natural sites (Table 1, type 1b). Only in one study also historical numerical data from pre-disturbance conditions were available (*Chovanec et al., 2002*).

   b. The endpoint was derived from the situation when *a statistical significant difference* was observed between the indicator values of the different types of sites: This endpoint was utilised to express a rehabilitated site's deviation from degraded reference conditions.

   In most BA-studies, reference comparisons were made separately for the pre- and post-project monitoring. In some BA-studies, however, pre- and post-project data were first compared in a separate statistical analysis of both the rehabilitated and the reference site. Subsequently, the test results of the two analyses were compared and a conclusion concerning the project effectiveness was drawn.

2. Descriptive endpoints:

   The rehabilitated site's similarity with near-natural reference conditions or its deviation from degraded sites was identified graphically or by a mathematical term, such as percentage similarity. Generally, such endpoints were used when either no numerical reference data were available (e.g. descriptive conceptual models as theoretical references) or more complex spatial or temporal patterns were studied. In some cases, however, rehabilitation and reference data were compared descriptively despite the availability of numerical data.

In 8 BA-studies (44 %), isolated before-after comparisons within the rehabilitated site were carried out, both descriptively and statistically. In all but one of these studies reference data would have been available.

For 1 indicator evaluation, we could not identify the endpoint used, because the reference conditions concerned were described ambiguously. Within 17 studies (49 %) multiple endpoint types were used, i.e. the same indicator was utilised several times to compare the rehabilitated sites with different reference types or to apply both statistical and descriptive analyses. In 7 studies endpoints were exclusively defined by description.

It is demanded in the literature that the significance of the natural variability be adequately considered in the reference and endpoint definition, i.e. the success of rehabilitation should not be determined on the basis of an "all or nothing single endpoint" (Gore and Milner, 1990; Palmer *et al.*, 2005). As described above, many of the reviewed case studies followed this principle by sampling spatial and temporal replicates of both rehabilitation and reference sites. However, several analyses were restricted to a before-after comparison within the rehabilitated sites while the potential change at simultaneously sampled reference sites remained unconsidered. As the degraded pre-project situation does not represent a reference condition, caution must be exercised in attributing the observed change to the rehabilitation measures (Chapman, 1999; Smith *et al.*, 1993; Underwood, 1994).

For spatially unreplicated (BACI-) studies, Conquest (2000) recommended the use of graphical methods instead of statistical tests. Resh *et al.* (1988) emphasize that not only endpoints but also response mechanisms should be studied. This implies, on the one hand, that relevant functional indicators be used (see paragraph 4.2). On the other hand, the temporal trajectory of the recovery needs to be considered (Gore and Milner, 1990). This aspect was hardly addressed in several studies.

5. Discussion of monitoring results

As noted above, our discussions are based on the investigators' conclusion on rehabilitation outcome. Recovery occurs over time. Therefore, the responses that might be observed are always a product of the study's duration and timing (Table 6).

5.1 General outcome

In all case studies at least one fish-biological endpoint was achieved. The proportion of positive results was highly variable. In 12 studies (34 %) positive outcomes prevailed. In 2 of these studies, even all the indicators displayed a positive development, e.g. in a restructuring project on a Coastal Oregon stream *(Crispin et al., 1993)*: During 3 post-project years, the average number of both coho salmon spawners (mean peak counts) and redds rose clearly at the treated sites whereas no such increase was observed in control systems nearby.

In 10 studies, positive and negative results occurred with equal frequency, while in 13 studies (37 %) negative results dominated. In a PT comparison of 13 British lowland rivers most of the indicators used revealed no significant difference between the rehabilitated sites -artificial riffles or flow deflectors- and the degraded reference sites 4 to 9 years after project implementation (*Pretty et al., 2003*). Only the two most abundant fish species, i.e. bullhead *Cottus gobio* and stone loach *Barbatula barbatula*, displayed significantly higher densities at the sites restructured with artificial riffles. *Roni (2003)* investigated the structure of the benthic fish fauna in 29 Pacific Northwest streams after the placement of woody debris. He found no significant difference between the rehabilitated sites and the untreated controls 1 to 12 years after project implementation.

In many studies, including the two aforementioned articles, fish-biological and abiotic metrics were directly compared to help explain the observed outcome. *Pretty et al. (2003)* analysed the percentage difference in biotic and abiotic indicators between rehabilitated sites and controls to test whether the stretches with the largest abiotic alteration performed also the highest biotic change. They found few significant relationships, such as the positive correlation of the mean flow velocity with the species richness and diversity. In a similar comparison, *Roni (2003)* found that juvenile lampreys (*Entosphenus tridentatus* and *Lampetra* spp.) and 1+ and older reticulate sculpins (*Cottus perplexus*) increased in density and size with the largest changes in the availability of large woody debris (LWD).

In 2 case studies, indicator values at the rehabilitated sites even exceeded those at the near-natural reference sites. *Paller et al. (2000)*, for instance, explained the elevated species richness, abundance and trophic diversity with the intermediate successional stage.

5.2 Spatio-temporal scale of recovery

Recovery rates of 1 month to 9.5 years were reported. In some cases, such as for the latter figure, these values represent conservative estimates as the combined analysis of all the post-project values hampered the precise determination of recovery rates.

In many studies, fish responded very rapidly to the physical habitat change, i.e. they displayed recovery rates ≤ 1 year: Following the placement of boulder clusters and V-dams in a Newfoundland stream, the 0+ and 1+ density of Atlantic salmon increased significantly within the first post-project year *(van Zyll de Jong et al., 1997)* at the rehabilitated site compared to the control. Compared to the pre-project situation the number of species at a restructured site of the Austrian Melk river rose from 10 to 16 within 12 months while it remained fairly constant at a canalised control site (*Jungwirth et al., 1995*). *Zika and Peter (2002)* placed LWD in a stream in Fürstentum Lichtenstein, Western Europe, and found significantly higher abundances of brown trout (*Salmo trutta fario*) and non-native rainbow trout (*Oncorhynchus mykiss*) in the treated section than in the control section after 86 and 29 days, respectively.

5.3 Explaining the outcome

Various factors were suggested in the case studies as affecting the overall recovery of the fish assemblages and thus the rehabilitation outcome.

Type of rehabilitation scheme: In 2 studies, authors critically question the appropriateness of the selected instream structures to enable recovery of the native fish fauna. The addition of flow deflectors and artificial riffles might have been inappropriate for the studied British low-gradient rivers (*Pretty et al., 2003*) where the creation of off-channel and marginal habitats might be a better strategy. *Lepori et al. (2005)* admitted that the placement of small relicts of formerly blasted rocks in Swedish streams possibly led to over-structured habitat conditions. In several studies the selected instream structures were referred to as 'interim management tools' to improve degraded conditions until the real causes of impairment could be eliminated.

Ongoing deficits in physical habitat: Limited biotic recovery might be explained by physical habitat characteristics that have not yet reached their desired conditions *(Raborn and Schramm, 2003)*. This might be a consequence of too short a time span for habitat adjustment. Furthermore, processes operating at a higher spatio-temporal scale might still be interrupted and continue to influence the local recovery at the rehabilitation sites. Through incision, original floodplains are completely isolated from the active channel thereby inhibiting long term recovery of the stream ecosystem *(Shields et al., 1998a)*. Low quantities of woody debris may limit pool availability and hence habitat quality for pool-dwelling species and age classes *(House, 1996)*.

Due to the presence of highly competitive riparian plants, such as the kudzu vine (*Pueraria lobata*) riparian processes and functions may be significantly altered *(Shields et al., 1997)*. In the lower river Rhine, the lack of suitable upstream spawning areas constrains the recovery of rheophilic species *(Grift, 2001)*.

Degree of ecosystem degradation: The extent of the initial degradation of the physical habitat may affect rehabilitation outcome. As discussed by *Shields et al. (1997)*, recovery at heavily degraded sites may take longer than at less impaired sites. This must not necessarily lead to limited project success as *Roni and Quinn (2001)* demonstrated in their analysis of LWD placements. About 3 to 7 years after project implementation they detected the largest physical and biological response at those sites that had low levels of LWD prior to rehabilitation. Similarly, smaller or no biotic responses are to be expected following rehabilitation in minimally degraded streams (*Rosi-Marshall et al., 2006*).

Sources of colonisers: Recovery was significantly influenced by the proximity, quality and connectivity of recolonisation sources, such as near-natural river sections or tributaries. This seemed particularly true for isolated small-scale rehabilitation measures. *Shields et al. (1998a)* reported slower recovery rates in rehabilitated reaches at greater distance from an undisturbed stream than in those closer to the recolonisation source. Limited recovery in 13 lowland rivers in Britain might be partly explained by the impoverished source assemblages in the study region *(Pretty et al., 2003)*. Open dispersal ways at catchment scale, i.e. a sufficient longitudinal connectivity, are the basic requirement for any immigration of potential colonisers *(Gowan and Fausch, 1996)*. Differences in accessibility might be responsible for the differing response of fish assemblages in several rehabilitated streams in Mississippi *(Shields et al., 1995; Shields et al., 1997; Shields et al., 1998a)*. The positive development of fish assemblages in restructured shoreline habitats of the Austrian Danube *(Chovanec et al., 2002)* may be related to the functioning bypass system at a nearby hydroelectric dam. However, well-connected nearby sources of colonisers did not generally lead to rapid or full recovery: *Lepori et al. (2005)* and *Raborn and Schramm (2003)* did not detect any differences between rehabilitated and control sites despite close proximity of a recolonisation source.

Sufficient lateral connectivity: Floodplain waters with moderate flow offer valuable nursery areas for juvenile rheophilic fish *(Chovanec et al., 2002; Grift et al., 2001)*. As larvae passively drift in from upstream spawning grounds *(Grift, 2001)*, the timing and duration of the connection with the main channel are crucial *(Chovanec et al., 2002; Grift et al., 2003)*.

Colonisation period and monitoring duration: Shields et al. (1995) explained the modest fish response to stone weir placement - among other factors - by the relatively short duration of the post-project monitoring of 1 year. In their analysis of 30 sites rehabilitated 4 to 9 years before, *Pretty et al. (2003)* found no significant relationship between the age of the schemes and the biotic parameters studied. Given the close proximity of recolonisation sources, *Raborn and Schramm (2003)* and *Lepori et al. (2005)* suggested that the colonisation period of 2 and 3 to 8 years, respectively, may not be responsible for the lack of positive response in the fish assemblage. *Brooks et al. (2004)* and *Rosi-Marshall et al. (2006)* argued that their post-monitoring of 0.5 and 1 to 3 years, respectively, was too short to detect real changes in the fish assemblage.

Availability of refugia: The positive impact of refugia on recovery was demonstrated by the different functions which they fulfilled. First, refugia created within the rehabilitated reaches were of direct benefit to fish by providing shelter from adverse conditions, such as natural extreme events (*Shields et al., 1998a*) or ongoing human disturbances *(Chovanec et al., 2002)*. Second, they may have indirect positive effects on fish, e.g. by improving food availability *(Langler and Smith, 2001)*.

Water quality: One case study discussed the significance of impaired water quality on the recovery process *(Pretty et al., 2003)*. The observed nutrient enrichment might have contributed to, but not solely caused the limited assemblage recovery.

Absorptive capacity: Rehabilitated river reaches may attract animals from nearby untreated reaches. *Gowan and Fausch (1996)* proved that the increased densities of adult salmonids in rehabilitated reaches of 6 Colorado streams resulted from a risen immigration. After LWD placement in a coastal Washington stream, *Cederholm et al. (1997)* observed increased winter retention of juvenile coho salmon. In both studies, authors weighted the results positively although the increases were not the result of enhanced in-situ processes, such as recruitment. Immigrants must have left suitable habitat that could afterwards be filled by subordinate conspecifics, thereby increasing their survival. An increased immigration might also be responsible for the rise in abundance observed in the Australian Williams river after reintroduction of wood *(Brooks et al., 2004)*. These results coincide with those from Niemi *et al.* (1990) who found that the recovery of fish populations following pulse disturbances was mainly driven by immigration and recolonisation and not by an increase in the resident populations.

Stocking with hatchery-reared fish: In 3 of the reviewed case studies, hatchery-reared individuals of native fish species were released, either in an attempt to artificially support and accelerate post-rehabilitation recovery of fish assemblages (2 articles) or to study their behaviour in a restructured river section (1 article). *Cederholm et al. (1997)* reported on the release of fed coho salmon fry during 3 of totally 6 post-project years after LWD placement. The comparison of stocked versus unstocked years revealed "no discernible effect on density of fish". In response to the low number of adult returns, coho salmon fry was released into the restructured Fish Creek, Oregon *(Reeves et al., 1997)*. After finding the stocked fry of one year infected with bacterial kidney disease, the ponds had to be dried out. Additional stocking 3 years later showed poor survival, so stocking was given up. To test the suitability of rehabilitated physical habitat for grayling (*Thymallus thymalus*), *Vehanen et al. (2000)* released twelve adult grayling, tagged with transmitters, into a Finnish river affected by morphological modification and hydropower production. The fish preferred the enhanced sites. Relatively high fishing pressure led to the loss of 5 out of 12 adults during the 30-day study period and another 2 after study end.

Timing of rehabilitation measures: Early responses of fish on physical habitat rehabilitation might be influenced by the season in which the measures were implemented, in particular where seasonal habitats are concerned. This is demonstrated in the increased use of restructured side-channels by juvenile coho salmon during winter months *(Giannico and Hinch, 2003)*.

The observations discussed above illustrate that the recovery of fish assemblages may depend on a variety of different factors. Hereby, it becomes apparent that the mitigation of certain stressors, such as physical habitat impairment, not necessarily involves the recovery of the biological components addressed. Similar results are reported from the field of ecotoxicology (Depledge, 1999).

6. Analysing the influence of the monitoring scheme on the project outcome

Different elements of a monitoring scheme may affect the outcome of a survey. In addition to the points mentioned by the authors of the case studies (paragraph 5.3), we analyse monitoring-specific subjects in the following sections.

6.1 Reference types

In 9 studies multiple references were applied (Table 5). Did the investigators come to different conclusions concerning the rehabilitation outcome depending on the type of reference used?

To account for indicator-specific characteristics which may also influence the results (hierarchical level, ecosystem attribute) we focused only on those indicators, which were analysed in multiple reference comparisons. Unfortunately, in none of the nine studies all the possible reference comparisons were made for all the indicators considered; in 2 cases even no indicator was compared against multiple references. The following discussion is based on the limited information available.

In 4 studies, multiple reference comparisons led to consistently positive or negative results in terms of endpoint achievement. In the single case, where post-project data were not combined or pooled, recovery rates differed depending on the reference type used: *Jungwirth et al. (1995)* descriptively compared the total number of species at a restructured site of the Austrian Melk river with both a canalised control site and a theoretical near-natural reference, i.e. a regression model linking the variability of the maximal depth to the species number. In the first reference comparison (control), the development was positively assessed already after 12 months (see section 5.2). The near-natural value predicted by the regression model was achieved only later, as was demonstrated by a survey 3 years after project end.

In 3 studies the rehabilitation outcome differed depending on the type of reference comparison, i.e. a positive outcome resulted from one reference comparison while for the other the endpoint was not achieved. On the one hand, this led to a critical discussion on the actual degree of system impairment, such as in a PT-comparison of rehabilitation projects in 7 Swedish streams *(Lepori et al., 2005)*: For most fish parameters there was no significant difference between near-natural and rehabilitated sites, although the latter did not differ significantly from the degraded

reference sites either. On the other hand, consideration of a further reference type offered the possibility to detect additional, more complex aspects of recovery: For an incised warmwater stream in Mississippi, *Shields et al. (1998a)* opposed the relative numerical and biomass abundances of the most dominant species at rehabilitated sites (woody vegetation, stone structures) with those of both a near-natural and a degraded reference site. In neither case were the statistic-based endpoints achieved after 3.5 years. The lot of the observed trends proved to be consistent with the predictions made by a third, theoretical reference, i.e. a conceptual model of fish community structure in small warmwater streams.
Despite data limitations which prevented us from identifying specific trends, the case-specific findings indicated that reference types used may influence the evaluation of the rehabilitation outcome.

6.2 Indicator characteristics

In a next step, we examined to what extent the observed results might be influenced by the indicator type used (Table 4). For instance, we were interested if recovery at a higher hierarchical level, such as the community, is generally accompanied by a recovery at a lower, e.g. population, level. To exclude possible impacts of multiple reference comparisons (see previous paragraph), we only considered those 25 studies that worked with a single reference type (Table 5).
Firstly, we discussed a possible effect of the indicator's hierarchical level on the recovery rate. For this purpose, we compared those studies that treated different hierarchical levels for a certain ecosystem attribute, i.e. the outcome of the three indicator types within each column of Table 4 was analysed. Seven studies met this assumption. Secondly, we checked for an effect of the ecological attribute that is addressed. For this reason, we analysed those studies that contained indicators from different attributes for a given hierarchical level, i.e. the outcome of the three indicator types within each row in Table 4 was compared. Eight studies could be considered.
In all but 3 studies, conclusions on project effectiveness were highly variable within the different indicator types (see below), i.e. different indicators addressing the same hierarchical level and ecosystem attribute provided different results. This complicated an overall comparison between different indicator types.

Two of the three studies that lacked such variability, showed homogeneously negative results for the different indicator types (see also paragraph 5.1 for the same two examples): *Pretty et al. (2003)* found no significant difference between control sites and sites enhanced by deflectors, neither in the population (total abundance of selected species) nor in the community composition (species richness, diversity, evenness, total abundance). In an evaluation of LWD projects, *Roni (2003)* detected no significant difference from the control sites for the population and the community structure of benthic fish and lampreys (abundance and length of different age classes of species and families, respectively). The third study with no variability within the different indicator types showed heterogeneous results between the hierarchical levels that were addressed: In their comparison of manipulated and untreated control sites in a British lowland river, *Langler and Smith (2001)* observed a significant change in community structure (abundance of 0+ fish) 18 months after project completion. In contrast, the population structure, i.e. the average length of the 0+ fish of selected species, showed no change over the same time period.
All ecosystem attributes within the population level were investigated in one monitoring study on a Norwegian stream restructured by weirs, pools and flow deflectors *(Linlokken, 1997)*. Homogeneously positive results were achieved in terms of population composition and function (abundance of brown trout and production and growth rate, respectively), whereas the structural indicators displayed variable outcomes. The average length within different age classes at treatment and near-natural reference sites did not differ neither in the pre- nor in the post-project monitoring. In contrast, graphical analysis of the age structure revealed heterogeneous results in the individual post-project years, attributable to the limited durability of the rehabilitation structures (see paragraph 5.3).
Such a variability of the results within one or several indicator types was observed in all the other studies that used a single reference type. Possible reasons are given below.

Sensitivity of the indicator: Paller et al. (2000) used different indicators to describe community composition in South Carolinian streams recovering from riparian degradation. In the analysis of the IBI only one variable, i.e. the percent tolerant fish, classified the replanted reaches as less natural than the near-natural reference sites, while the overall index indicated that the community composition has recovered.

In contrast, the multivariate statistical analysis of the absolute species abundances revealed differences between the two types of sites due to higher densities of certain species. Based on previous work, the authors conclude that the IBI may be particularly sensitive for the early stages of recovery ($\leq$ 2 years) while in a later phase the multivariate methods would be more sensitive.

Different response of individual species or age classes: In several studies, different species or age classes reacted differently to physical habitat rehabilitation, e.g. when species- or age-class-specific habitat preferences are met. *Gowan and Fausch (1996)* found a significant increase in abundance and biomass of adult trout, but not juveniles, in rehabilitated reaches of 6 Colorado streams relative to controls. The observed pattern could be explained by the functional indicator 'fish movement', displaying an increased immigration of adult animals from beyond the reach boundaries. Habitat preferences may vary between the seasons. Accordingly, *Roni and Quinn (2001)* found a stronger response for juvenile coho salmon density on LWD structures during winter compared to summer.
An observed pattern of abundance might also reflect biotic processes, such as predation or competition. In 2 articles it was supposed that the increased abundance of large trout at a rehabilitated site may have contributed to the synchronous decrease of younger conspecifics *(Kelly and Bracken, 1998; Rosi-Marshall et al., 2006)*. In contrast, the increased densities of coho salmon in LWD projects did not seem to have established at the expense of the benthic species of the community *(Roni and Quinn, 2001; Roni, 2003)*.

Heterogeneous results between spatial replicates: Some studies reported highly variable results from different rehabilitated sites. In a newly created meander on an Indiana stream, fish biomass and mean individual biomass had increased significantly already after 9 months compared to control sites *(Moerke and Lamberti, 2003)*. The values in the second meander remained at, or below, those of the control. *Giannico and Hinch (2003)* demonstrated the effect of site-specific conditions, such as the temperature regime, on the rehabilitation outcome. Wood placement in a surface-fed side channel in British Columbia led to an increase in juvenile coho winter abundance and spring smolt output compared to the untreated control. In a groundwater-fed side channel, however, the same treatment resulted in a slightly reduced juvenile winter abundance and smolt output.

Heterogeneous results between temporal replicates: In several studies, indicator results were not persistent over time. On the one hand, this was explained by a limited durability of the habitat structures that were built. *Linlokken (1997)* observed a rapid positive response of brown trout density to weirs and pools created in a small Norwegian stream while at a near-natural reference site densities displayed only minor fluctuations. After a heavy flood and a cold winter, pools were filled at the treatment site, and trout densities dropped back to pre-project values. At the near-natural reference site no such decrease was observed.

On the other hand, physical habitat alterations in the catchment might influence local performance. *Rosi-Marshall et al. (2006)* observed a two-fold increase in the abundance of harvestable trout (brook trout, *Salvelinus fontinalis*, and non-native brown trout) in a stream section in Northern Michigan treated by k-dams. After 3 years no difference was found between k-dam section and control. The authors assumed that this resulted from a higher attraction of cover-providing skyboom-structures which in the meantime were built nearby.

High heterogeneity within the individual indicator types prevented a general analysis of the possible interrelationships between different hierarchical levels or ecosystem attributes. In most studies, such correlations remained completely unconsidered. However, this would be imperative for an appropriate monitoring design as the spatio-temporal scale over which the individual measurements vary, may differ (Chapman, 1999).

Some findings in the literature indicate that response time may vary between different hierarchical levels or ecosystem attributes. In a general review on ecosystem recovery following pollution, Depledge (1999) stated that "both the rate and extent of recovery may differ at different levels of biological organisation". Furthermore, he showed that compositional parameters may fail to recover to pre-disturbance conditions, although ecosystem structure and function were reestablished. Detenbeck *et al.* (1992) reviewed the natural, unassisted recovery of temperate-stream fish communities following various natural and anthropogenic disturbances. Community parameters displayed considerably shorter recovery rates than population metrics. This might be explained by the low resilience of single fish species.

7. Summary and conclusions

Due to the uniqueness of the individual projects (Middleton, 1999), the sometimes limited data availability and the resulting heterogeneity of the data set, a generalisation of the findings was not possible. Despite this, implications for future monitoring and river management can be derived.

Studying the temporal trajectory of recovery: Individual stages in the recovery process of fish assemblages are still poorly understood (*Lepori et al., 2005*; Nilsson *et al.*, 2005a; Roni *et al.*, 2002). Only few studies specifically addressed the temporal trajectory of recovery. Rather, the results of different spatial replicates were analysed in combination or by pooling. Future monitoring projects should therefore increasingly focus on the temporal dimension of recovery in order to identify potential patterns of general validity. Over the entire recovery process, responsible mechanisms might vary (*Paller et al., 2000*) implying the use of different indicators at different stages, e.g. functional indicators for the early phases (Kelly and Harwell, 1990), followed by compositional and structural indicators. When applying different indicator types, the spatio-temporal scales of the individual indicators need to be considered (Chapman, 1999).

Monitoring ecosystem functions: Few functional indicators were used in the reviewed case studies. Among other reasons, this might be due to the lack of appropriate evaluation guidelines, i.e. the availability of practical, cost-effective indicators. This research gap needs to be filled to further improve understanding of important riverine processes and their effect on ecosystem composition and structure.

Including different indicator types: We showed that the investigators' conclusions on the rehabilitation outcome might vary strongly depending on which hierarchical level or ecosystem attribute was studied. To achieve a scientifically supported monitoring of the biological effectiveness, a sophisticated selection of many different indicator types is recommended (Angermeier, 1997). At the same time, this enables the study of functional relationships between different organisational levels of the riverine ecosystem. Certainly, success evaluation should not be based solely on fish indicators (Roper *et al.*, 1997), even though the project focussed mainly on the fish fauna.

Including different reference types: It must be considered in the project evaluation that different reference types vary in their quality and meaning. If rehabilitation data can not be compared with both reference and control sites, caution must be exercised in attributing the observed changes to the rehabilitation measure (Chapman, 1999). When multiple references are used, preferably all reference comparisons should be made for the entire indicator set. This enables a true comparison between different indicator types without the confounding effects of multiple reference types.

Spatial and temporal replication: Beside the consideration of reference information, adequate spatial and temporal replication is needed to accurately distinguish between rehabilitation-induced change and natural variability (Kelly and Harwell, 1990; Underwood, 1992). In addition, more long-term and large-scale studies are required to improve the understanding of the spatio-temporal development of systems following rehabilitation (Poff and Ward, 1990; Roper *et al.*, 1997).

Formulation of specific hypotheses: River rehabilitation and monitoring should increasingly follow an experimental approach where specific hypotheses are formulated and tested by appropriate statistical procedures (Block *et al.*, 2001; Chapman, 1999).Both practitioners and the scientific community would benefit from the insights gained, e.g. by an improvement of the still limited predictive capacity on the rehabilitation outcome.

Evaluating overall success: In all case studies, positive developments in one or several fish indicators were found. However, it is a challenging task to draw conclusions from these evaluations on the indicator level to the fish-specific effectiveness of the overall project. Project success depends on further factors, such as the specific aims of the projects and their weighting by the project managers (Cairns, 1990). Overall project effectiveness was not explicitly evaluated in any of the studies we reviewed. For this purpose, concrete, practical suggestions for evaluation are required.

The questions we addressed in this review would ideally be investigated using a meta-analysis (Arnqvist and Wooster, 1995), i.e. a statistical approach to summarise and compare multiple independent case studies. This would allow to objectively account for the different factors that may influence recovery. However, the limitations we faced concerning data availability, temporal resolution within the recovery trajectory and inconsistent reference comparisons complicate the use of such applications.

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Appendix

Table 8: Indicators used in the case studies to measure the recovery in fish assemblages. Only those indicators are given that were evaluated and discussed in the articles. In most studies several indicators were measured. The level within the ecological hierarchy (Angermeier, 1997) is indicated in superscript: $^{C)}$ community, $^{G)}$ guild, $^{P)}$ population level.

Compositional indicators	No. (%)	Structural indicators	No. (%)
Species richness $^{C)}$	11 (31)	Age structure	
Species diversity $^{C)}$	7 (20)	By taxonomic groups $^{C, P)}$	5 (14)
Species evenness $^{C)}$	5 (14)	Presence-absence of different age classes	
IBI $^{C)}$	1 (3)	By taxonomic groups $^{C, P)}$	3 (9)
Total abundance		By taxonomic-functional groups $^{G)}$	1 (3)
All Fish $^{C)}$	11 (31)	Total abundance of different age classes	
By taxonomic groups $^{C, P)}$	2 (6)	All fish $^{C)}$	3 (9)
Relative numerical abundance		By taxonomic groups $^{C, P)}$	11 (31)
By taxonomic groups $^{C)}$	8 (23)	By taxonomic-functional groups $^{G)}$	2 (6)
By guilds $^{C)}$	2 (6)	Total abundance of spawners	
Total biomass		By taxonomic groups $^{C, P)}$	2 (6)
All Fish $^{C)}$	8 (23)	Rel. numerical abundance of diff. age classes	
By taxonomic groups $^{C, P)}$	2 (6)	By functional groups $^{C)}$	1 (3)
Relative biomass abundance		By taxonomic groups $^{C)}$	1 (3)
By taxonomic groups $^{C, P)}$	2 (6)	Total biomass of different age classes	
		By taxonomic groups $^{C, P)}$	1 (3)
		Average individual biomass	
Functional indicators		All Fish $^{C)}$	1 (3)
Survival $^{P)}$	3 (9)	Average individual biomass of diff. age classes	
Growth rate $^{P)}$	3 (9)	By taxonomic groups $^{P)}$	1 (3)
Fish Movement $^{P)}$	3 (9)	Average fish length	
Production $^{P)}$	1 (3)	By taxonomic groups $^{C, P)}$	5 (14)
		Minimum fish length	
		By taxonomic groups $^{P)}$	1 (3)
		Maximum fish length	
		By taxonomic groups $^{P)}$	1 (3)
		Variation in fish length	
		By taxonomic groups $^{P)}$	1 (3)
		Density of redds	
		By taxonomic groups $^{C, P)}$	4 (11)

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CHAPTER 6

Synopsis

"Restoration is not only a problem-solving matter;
it is a tool for ecological research"
Jordan and others (1987)

For centuries rivers were exploited by man in various ways. These exploitations have led to considerable degradation of riverine ecosystems. Today, 59 % of the world's large river systems are moderately to strongly affected by flow regulation, caused e.g. by reservoir operation, irrigation or fragmentation by dams (Nilsson and others 2005). Running waters are among the most threatened ecosystems worldwide (Dudgeon and others 2006). Ecosystem services, such as the provision of drinking water resources, fishery and flood retention, are also considerably affected by the degradation of running water systems (Cowx and others 2004; Nienhuis and others 2002).

In recent years, increasing efforts have been undertaken to solve the existing problems by returning impaired river systems to a more natural state (restoration and rehabilitation, respectively; Bradshaw 1996). Applied measures address different elements of the riverine ecosystem, e.g. riparian vegetation, aquatic fauna or river-groundwater interaction. To be sustainable, rehabilitation projects should highlight environmental, societal and economic concerns equally (BWG 2001).

However, problem-solving is only one element of restoration (Jordan and others 1987). Beside it, restoration projects offer a great opportunity for scientists from various disciplines to test and refine their understanding of riverine processes and functions (Bradshaw 1987).

River ecology is a rather young science and many riverine patterns and processes are still poorly understood. Several of the most influential theoretical concepts in river ecology are based on data gained from regulated, ca-nalised rivers (Ward and others 2001). Rehabilitation projects therefore provide an excellent testing ground for ecological theory. And vice-versa, these experiences can be transferred back to the practice to further develop a technologically, ecologically and socially sustainable river management.

The ideal project procedure

In order to fulfil the expectations concerning the gained insights and results, rehabilitation projects must be well-designed. Ideally, the project procedure comprises five steps (Figure 1). The first step involves the formulation of a guiding image ('Leitbild') which describes the target conditions for rehabilitation (Jungwirth and others 2002; Palmer and others 2005). The monitoring of baseline data is imperative as it provides useful information on the system's state, i.e. on impaired ecosystem processes and factors limiting biotic production (Kauffman and others 1997; Kondolf and Downs 1996).

In a second step, an interdisciplinary planning team formulates the project objectives according to the previously established guiding image. This is a participative process in which conflicting interests of stakeholder groups need to be identified and integrated, e.g. by adequate decision support tools (Hostmann and others 2005a; Hostmann and others 2005b). By comparing different project alternatives, the appropriate rehabilitation measure is finally selected (Hostmann 2005). After detailed planning of the measure, a success evaluation programme is devised which equally considers ecological, societal and economic aspects (Block and others 2001; Roni and others 2005; Woolsey and others 2005). During the implementation phase, the rehabilitation measure is realised. Afterwards, an exploitation phase follows in which the rehabilitation outcome is evaluated and communicated to the public. Ongoing deficits can be addressed by adaptive management (Block and others 2001; Kershner 1997). Future projects may benefit from the lessons learned.

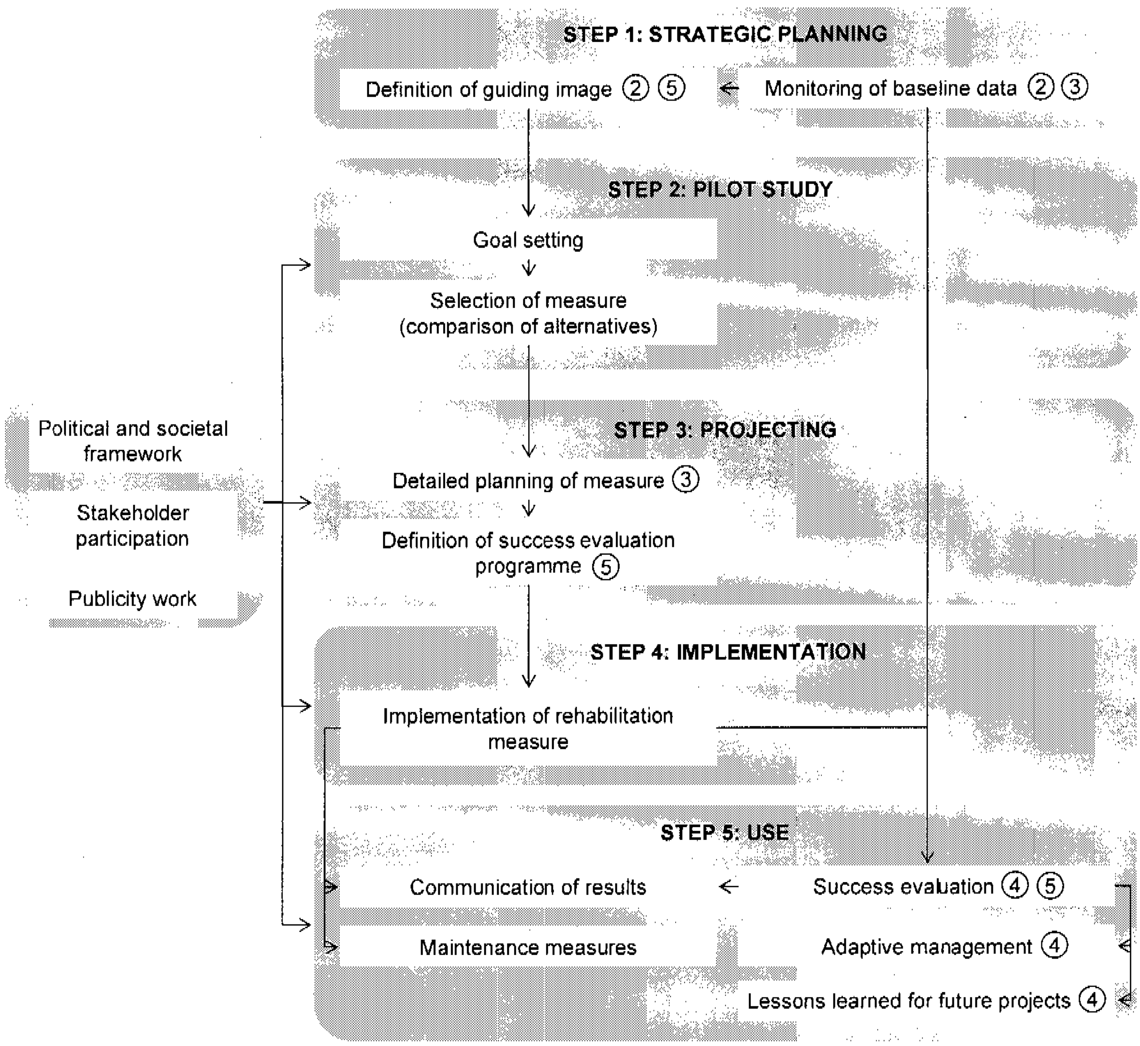

Figure 1: Elements of an ideal procedure for rehabilitation projects (Woolsey and others 2005, strongly modified from Holl and Cairns 1996). The numbers in circles indicate the chapters in this thesis which discuss aspects of the particular element.

In practice, however, important elements of the project procedure are often missing. In their worldwide review on the effectiveness of riverine habitat rehabilitation, Roni and others (2005) identified three common shortcomings. Firstly, rehabilitation schemes often lack a comprehensive assessment of historic and actual conditions before project start. Secondly, most measures are designed with the local level in mind, neglecting processes and deficits operating at watershed scale. Thirdly, a thoroughly planned, well-funded evaluation of the outcome is still rarely conducted. These conclusions are widely documented in the literature (e.g. Kondolf and Downs 1996; Kondolf 1995; Bash and Ryan 2002).

The findings reveal several gaps in the current management practice, e.g. between theoretical concepts and practical implementation or between baseline monitoring and success evaluation.

Insights from this thesis

In the present thesis, selected aspects of the rehabilitation procedure were investigated (Figure 1), focusing specifically on the evolution of the degradation and the assemblage recovery following rehabilitation. The results may contribute to filling some of the gaps identified before. In the following paragraphs, the findings from the individual studies are placed in the broader theoretical and practical context discussed above.

Consideration of near-natural reference conditions

The results gained in the individual studies demonstrate the special significance of near-natural reference conditions in the rehabilitation procedure, for both abiotic and biotic elements. Reference information may be obtained from different sources, such as historical documents, current undisturbed systems or theoretical concepts and models (Chapter 5, Palmer and others 2005). In this thesis we focused particularly on the use of historical reference information, such as old topographical maps and early inventories of fish species.

Near-natural reference conditions provide a valuable benchmark for determining the naturalness of a given site (Hohensinner and others 2003). Prior to rehabilitation, this enables the assessment of a system's actual degradation (Chapters 2 and 3), i.e. the extent of human modification of ecosystem structures and functions. A good knowledge of the present conditions and the evolution of deficits over time is an important prerequisite for the identification of the guiding image, the project objectives and adequate rehabilitation measures (Jungwirth and others 2002).

After rehabilitation, a comparison with the near-natural reference conditions supplies important information on project effectiveness (Chapters 4 and 5). A comparison with degraded reference reaches displays whether the treated sites developed differently than their untreated counterparts (effect of rehabilitation). A comparison with near-natural reference conditions, on the other hand, informs about the quality of that change (Chapman 1999). Furthermore, it may give clues to ongoing deficits, such as the still impaired temperature regime in the structurally rehabilitated river Thur (Chapter 4). Lessons learned might influence both further maintenance and management practices at a given rehabilitated site (adaptive management; Kershner 1997) and the planning of future rehabilitation projects. Ideally, a monitoring programme considers both degraded and near-natural reference conditions (Chapter 5; Chapman 1999).

Given the ongoing degradation of natural habitats (Balmford and Bond 2005) and the limited availability of financial, temporal and energy resources (Angermeier 1997), it is essential that priorities be set for restoration and conservation activities. Historical maps of an adequate quality and resolution can be used for the spatially explicit identification of priority sites for rehabilitation (Chapter 3). This method provides a helpful extension to existing approaches for prioritising rehabilitation activities (see e.g. Roni and others 2002).

Monitoring ecosystem functions

In many rehabilitation projects, lost habitat structures are actively recreated (Chapter 5), for example, through the placement of boulders (Lepori and others 2005), large woody debris (Hilderbrand and others 1997) or spawning gravel (Iversen and others 1993). Most of these measures emphasise individual, usually salmonid species (Frissel and Ralph 1998). Several studies have documented a limited durability of artificial habitat structures (Frissell and Nawa 1992; Linlokken 1997; Roni and others 2002), particularly in streams with high peak flow or high sediment load. In recent years rehabilitation has therefore increasingly focused on the recreation or maintenance of ecosystem processes and functions (Chapter 3; Angermeier and Karr 1994; Frissel and Ralph 1998; Muhar and Jungwirth 1998). A main goal is the reestablishment of dynamic habitat structures. In functioning ecosystems, site-specific habitats are naturally formed and connected (Beechie and Bolton 1999; Roni and others 2002), over various spatial and temporal scales.

Patches are continually destroyed and recreated by spatio-temporal variations in processes. In this way, a shifting mosaic of characteristic habitats is formed, which fulfils the requirements of different fish species and age classes (Bormann and Likens 1979). To evaluate whether the re-establishment of processes was achieved, functional indicators are needed which represent direct measures of ecological and evolutionary processes (Noss 1990). Functional metrics might be of special importance in the early stages of recovery as they directly indicate the presence of ongoing or new stresses ("early warning indicators"; Kelly and Harwell 1990). Furthermore, functional metrics permit to monitor specifically biological response mechanisms (Resh and others 1988) which might enable a more comprehensive interpretation of the project results (Chapter 5; Gowan and Fausch 1996). However, functional indicators are still highly underrepresented in today's monitoring studies (Chapter 5). Among other reasons, this might result from the lack of appropriate evaluation guidelines, i.e. the availability of practical, cost-effective indicators. This research gap needs to be filled in order to further improve the understanding of important riverine processes (Ward and others 2001), such as the resilience of biotic communities following rehabilitation.

Additionally, monitoring should increasingly utilise the potential of experimental design (Chapter 5). This includes the formulation and testing of specific hypotheses on the expected project outcome (Block and others 2001). In order to delineate between natural variability and rehabilitation-induced change, appropriate spatial and temporal replication is needed (Underwood 1992). A wider spread use of experimental design could help to further improve the limited predictive capacity on rehabilitation outcome (Chapter 5).

In retrospect, a greater emphasis on ecosystem functions would have been valuable also in our own studies. Indeed, structural and compositional indicators may give clues on ecosystem functions (Noss 1990). In the rehabilitated reaches of the river Thur, for instance, we observed an elevated abundance of backwater habitats relative to canalised sections (Chapter 4). Fish assemblages may therefore display an increased resistance to, and resilience from, natural disturbances (Lusk and others 2001; Sedell and others 1990; Townsend and others 1997). However, the presence of such refugia is only an indirect measure of ecosystem resilience which does not prove the backwater's ecological functioning.

Spatio-temporal scale of degradation and rehabilitation

Considering a broader spatio-temporal scale is of special importance in riverine systems that suffer from multiple degradations (Kondolf and Downs 1996). For instance, habitat rehabilitation under impaired hydrological conditions as prevail in the Swiss river Rhone (Chapters 2 and 3) is particularly delicate and challenging (Unfer and others 2004). In most cases, morphological measures alone will not suffice to compensate for hydrological deficits (Fette and others in press). Hydrological actions, such as the mitigation of hydropeaking, are also needed to facilitate the system's recovery. For achieving this, various technical and operational measures are suggested in the literature. These measures include e.g. the continuous discharge from retention basins or slower ramping rates of the turbines (Halleraker and others 2003; Moog 1993).

In general, the public takes a favourable view on river rehabilitation (Junker and Buchecker, submitted). Rehabilitated rivers and streams are considered attractive recreation areas. In many cases, biological effectiveness is automatically inferred from scenic attractiveness.

The failure of numerous projects, however, demonstrates that many aspects of ecosystem rehabilitation and recovery are not yet fully understood. Great efforts are needed in order to improve future rehabilitation practice and to maintain public goodwill for rehabilitation projects.

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