ZUM LEISTUNGSVERMÖGEN AUSGEWÄHLTER METHODEN ZUR BEWERTUNG DER ÖKOLOGISCHEN FUNKTIONSFÄHIGKEIT VON FLIESSGEWÄSSERN

The efficiency of selected methods to assess the ecological integrity of running waters

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Vorbemerkungen zum Aufbau der Dissertation

Vorliegende Dissertation besteht im Wesentlichen aus zwei Teilen, einer Einleitung mit folgender Zusammenfassung und kurzer Diskussion der wesentlichen Methoden und Ergebnisse aus acht Publikationen. Den zweiten Teil bilden diese acht Publikationen selbst, welche in Englischer Sprache verfasst sind und international als positiv begutachtet wurden. Diese sind thematisch gereiht und werden in weiterer Folge mittels Nummern in eckigen Klammern [ ] zitiert.

Sechs Artikel [2, 3, 4, 5, 6 und 8] wurden bereits in den Jahren 2006 und 2007 in SCI Journalen veröffentlicht. Einer erschien dieses Jahr in einem Konferenzband [7] und ein weiterer wurde diesen Sommer in einem „peer review“ Journal [1] angenommen.

Der Reihung der Artikel liegt weder der Zeitpunkt der Veröffentlichung noch eine wissenschaftliche Gewichtung zugrunde. Vielmehr umspannen sie einen breiten Bogen von der Grundlagenforschung über standardisierte Bewertungsmethoden bis hin zu Charakteristiken von unterschiedlichen Belastungen an Fließgewässern.

Preliminary remarks on the structure of the thesis

This PhD thesis consists of two parts: an introductory text with the summary of the main results and eight articles, which constitute the core of the thesis. To date, seven articles have been published; one manuscript has been accepted and submitted for publication.

The summary comprises the key statements of the thesis. Detailed data and information are referred to the individual articles. These are cited in the text by means of numbers in square brackets [ ].

INHALTSVERZEICHNIS

Kurzfassung

Im Jahr 2000 führte die Europäische Union (EU) die Wasserrahmenrichtlinie (WRRL) ein, eines der weltweit modernsten Gesetze zum Thema Wasser, dessen vorrangiges Ziel es ist, bis zum Jahr 2015 den „guten ökologischen" Zustand von Fließgewässern zu erreichen. Fische sind als besonders sensibel gegenüber anthropogenen Eingriffen (Hydrologie, Morphologie, Kontinuum und Wasserqualität) bekannt, deswegen wurden sie für die WRRL als mögliche Indikatoren, welche den ökologischen Zustand anzeigen können, ausgewählt. Ein wesentliches Ziel dieser Dissertation ist daher die Entwicklung und Evaluierung einer geeigneten Methode zur Bewertung der ökologischen Funktionsfähigkeit von Fließgewässern.

Um den aktuellen Zustand der europäischen Fischfauna unter Berücksichtigung von verschiedenen Belastungen analysieren zu können, wurde zunächst eine Datenbank mit mehr als 8 200 Probenstellen samt Fisch-, Umwelt- und Belastungsdaten zusammengestellt. Zusätzlich wurden relevante Kriterien definiert, die Art und den Grad der anthropogenen Belastungen charakterisieren. Zwei unterschiedliche Ansätze zur Bewertung wurden schließlich gewählt: die so genannten „spatially based" Modelle (SBM EU) und ein multimetrisches Modell, dem später weiterentwickelten Europäischen Fischindex (EFI, Pont et al. 2007). Ein Vorteil des EFI ist, dass er, obwohl aus einem Index bestehend, in weiten Bereichen anwendbar ist, was eine Vergleichbarkeit von Bewertungsergebnissen innerhalb der EU ermöglichen soll.

Multivariate Analyseverfahren von verschiedenen Belastungstypen, samt der neuerdings zu berücksichtigenden Klimaerwärmung, auf verschiedenen räumlichen und zeitlichen Betrachtungsebenen, erlauben die Quantifizierung des Zusammenhanges von Fisch und Eingriff. Zukünftige Forschungen sollen vor allem den kumulativen Einfluss von Belastungen beinhalten. Zusätzlich unterstützen Habitatmodellierungen die Auswahl und Beurteilung von geeigneten Restaurationsmaßnahmen, um, wie von der WRRL gefordert, den guten ökologischen Zustand von Fließgewässern zu erreichen.

Diese Arbeit wurde hauptsächlich im Rahmen von FAME (http://fame.boku.ac.at), einem Projekt des fünften Rahmenprogramms der EU, erstellt.

ABSTRACT

In the year 2000, the most modern water legislation in the world – the Water Framework Directive (WFD) – was launched by the European Union. One of its key objectives is to achieve the “good ecological status” of running waters within the next 15 years. As fish are known to be sensitive to chemical and hydromorphological pressures, in particular continuum interruptions, the WFD consider them as potential indicators for assessing the ecological status of rivers. Therefore, the objectives of this PhD thesis were to develop and evaluate a standardised fish-based method for assessing the ecological status of European running waters.

In order to analyse the present status of the European fish fauna with regard to typical chemical and hydromorphological pressures, a database containing information on more than 8,200 fishing sites was compiled. Relevant criteria characterising anthropogenic pressures such as hydrological alterations (flow diversion, hydropeaking), impoundment, channelisation, continuum interruptions (lateral, longitudinal) and water quality changes were defined.

Two different methodologies were used: the so-called spatially based modelling (SBM EU) and multi-metric modelling, the latter leading to the European Fish Index (EFI, Pont et al. 2007). Despite being a single index the EFI is applicable to a wide range of environmental conditions across Europe and is suitable for intercalibration purposes.

Multivariate analyses of different types of pressures including global warming on different spatial and temporal scales permit a quantification of relationships between pressures and fish. Future publications should focus on the cumulative impact of pressures. Additionally, habitat modelling results should be used to support the selection of the most appropriate combination of restoration measures, according to the WFD, to achieve good ecological status or potential of running waters.

This thesis was carried out mainly within FAME (http://fame.boku.ac.at), a project under the fifth R&D Framework Programme of the European Commission.

EINLEITUNG

Die europäischen Fließgewässer sind vielfältigen anthropogenen Beeinträchtigungen ausgesetzt. Während die Gewässergüte in vielen Ländern nicht mehr zu den prioritären Problemen zählt, weisen die meisten Gewässer hydromorphologische Veränderungen auf. Zur Wiederherstellung des „guten ökologischen Zustandes“ bzw. des „guten ökologischen Potentials“ im Sinne der EU Wasserrahmenrichtlinie (Amtsblatt der Europäischen Gemeinschaften 2000) sind daher für diese Gewässer zunächst geeignete Methoden zur Bewertung des ökologischen Zustandes zu entwickeln.

Fische spielen aufgrund ihrer besonderen Indikatorfunktion schon seit langem eine wesentliche Rolle in der Bewertung des ökologischen Zustandes von Fließgewässern. Die Bedeutung von Fischen als Indikatoren wurde in den letzten Jahrzehnten zunehmend erkannt und diskutiert. Fische reagieren auf Veränderungen des Lebensraumes, auf Eingriffe in das Gewässerkontinuum und auf die Abtrennung der natürlichen Überflutungsräume. Sie eignen sich damit hervorragend für einen integrativen Bewertungsansatz, der sowohl für eine Beurteilung der menschlichen Einflüsse als auch für die Erfolgskontrolle von Schutz- und Revitalisierungsmaßnahmen unerlässlich ist.

Wie oben erwähnt sind Fische aus mehreren Gründen ausgezeichnete Indikatoren, um den ökologischen Zustand von Fließgewässern anzuzeigen (z.B. Jungwirth et al 2003):

- sie kommen in fast jedem Gewässer vor;
- ihre Lebensweise und die Ansprüche an die Lebensräume sind besser bekannt als bei anderen Organismen;
- sie zeigen aufgrund ihrer Bindung an unterschiedliche Habitate die Habitatqualität auf verschiedenen räumlichen Ebenen;
- sie zeigen aufgrund von Wanderungen zwischen verschiedenen Habitaten die Vernetzung im Längsverlauf („Gewässerkontinuum“) sowie jene mit dem Umland („laterale Konnektivität“);
- sie decken in der Nahrungspyramide unterschiedliche Ebenen von pflanzenfressenden Primärproduzenten bis hin zu Raubfischen ab;
- sie sind langlebige Organismen und zeigen Veränderungen und Entwicklungen über einen dementsprechend langen Zeitraum an;
- es liegen historische Informationen über die Verbreitung vor, und

- sie haben sowohl ökonomischen als auch ästhetischen Wert, und steigern somit das Bewusstsein für die Notwendigkeit der Erhaltung und Wiederherstellung aquatischer Ökosysteme.

Aus einer Vielzahl fallspezifischer Einzeluntersuchungen ist bekannt, dass Fische sehr gut hydromorphologische und chemische Belastungen anzeigen. Dem steht jedoch ein deutlicher Mangel an geeigneten, standardisierten Bewertungsmethoden gegenüber.

Bislang wurden jedoch erst wenige Versuche unternommen, anhand eines großen Datensatzes die Wirkungszusammenhänge zwischen anthropogenen Belastungen und Fischen systematisch zu untersuchen und die daraus abgeleiteten Erkenntnisse zur Entwicklung von Bewertungsmethoden heranzuziehen.

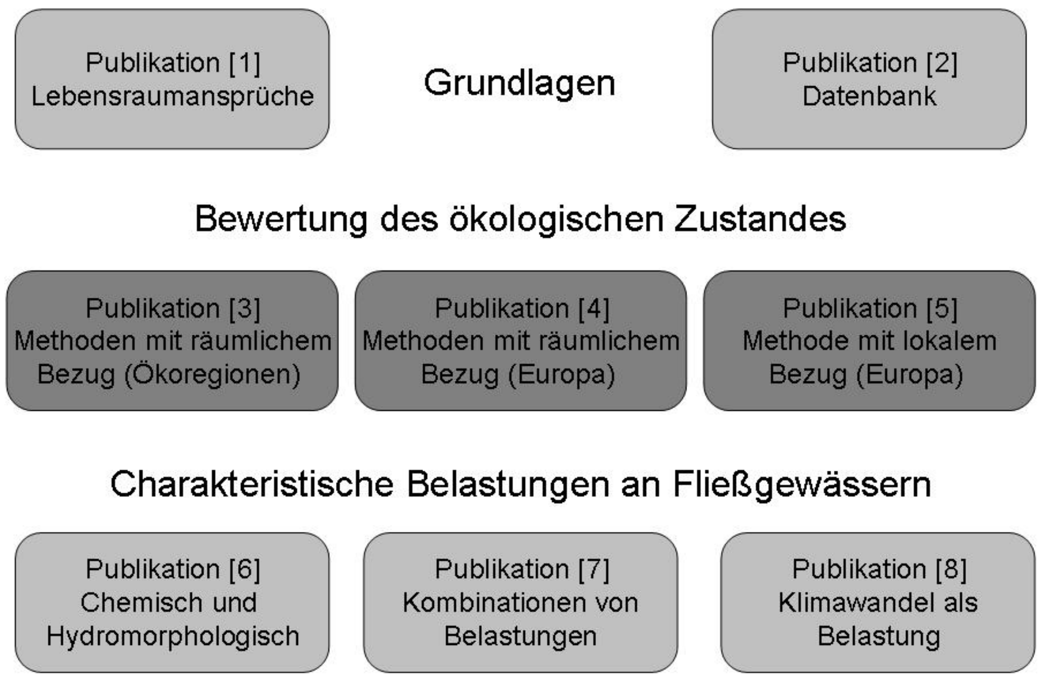

Abbildung 1: Grafisches Konzept aller acht Publikationen unter Berücksichtigung ihrer Thematik.

Ziel vorliegender Dissertation ist einerseits die Erarbeitung von Grundlagen für die Entwicklung von fischbezogenen Bewertungsmethoden und Restaurationsmaßnahmen und andererseits die Entwicklung von Bewertungsverfahren für eine flächendeckende Bewertung europäischer Fließgewässer. Dieses Konzept geht davon aus, dass der Zustand eines Gewässers als graduelle Abweichung von einem unbeeinträchtigten Referenzzustand definiert wird. Dieses Grundprinzip geht zum einen auf den von Karr 1981 in den USA entwickelten IBI (Index of Biotic Integrity) und auf das österreichische Konzept zur Bewertung der ökologischen Funktionsfähigkeit zurück (Schmutz et al. 1999 und 2000).

Aufgrund der thematischen Anordnung (Abbildung 1) und des Inhaltes der Publikationen ergeben sich folgende Problemstellungen, die im Rahmen dieser Dissertation aus ökologischer und statistischer Sicht aufgearbeitet werden:

1. Wie lassen sich Lebensraumansprüche von Fischen mit Hilfe von uni- und multivariaten Methoden darstellen? [1]
2. Kann eine hierarchische Datenbank sowohl Fisch-, Umwelt- als auch Belastungsdaten enthalten? Welche Voraussetzungen sind bei der Datenbankerstellung zu beachten? [2]
3. Eignet sich die deskriptive Statistik um Datenfehler (z.B. Ausreißer, missing data) zu identifizieren? [2]
4. Gibt es charakteristische Fischartenvergesellschaftungen in Europa, und wie lassen sich diese definieren? [3, 4]
5. Welche statistischen Verfahren sind geeignet, um Fischartenvergesellschaftungen einerseits für Ökoregionen und anderseits für einen ganzen Kontinent zu identifizieren? [3, 4]
6. Wie können Fischartenvergesellschaftungen mit Hilfe von Umweltvariablen (z.B. Seehöhe, Gewässerbreite) für beliebige Gewässerabschnitte prognostiziert werden? [4]
7. Lässt sich die natürliche Variabilität des Fischvorkommens mit Hilfe von Umweltvariablen erklären? [2, 3, 4, 5]
8. Welche Metrics (Kennziffern wie die Dichte von Kieslaichern) eignen sich besonders, um den Grad von anthropogenen Eingriffen widerzuspiegeln? [3, 4, 5]
9. Welche statistischen Verfahren sind geeignet, um standardisierte Bewertungsmethoden für Ökoregionen und Europa zu entwickeln? [3, 4, 5]
10. Welche Arten von Belastungen und Belastungsstufen können in europäischen Flüssen auftreten und wie häufig sind diese? [6, 7]
11. Gibt es andere zusätzliche Belastungstypen (z.B. Landnutzung oder Klimaerwärmung), welche bisher bei der Bewertung von Fließgewässern, zumindest in Europa, kaum berücksichtigt wurden? [8]

Diese Arbeit stellt einen umfassenden Versuch dar, eine von menschlichen Eingriffen weitgehend beeinflusste Flusslandschaft auf Basis von Fischen zu bewerten, sowie in weiterer Folge Eingriffe und Belastungen auf Fluss-Ökosysteme zu analysieren. In nächster Zukunft gilt es nunmehr, die gewonnen Erfahrungen zu verfeinern bzw. weiterzuentwickeln.

PUBLIKATIONEN IM ÜBERBLICK

[1] Melcher A. & Schmutz S. (2009): **The importance of structural riverine features of spawning habitat of nase *Chondrostoma nasus* (L.) and barbel *Barbus barbus* (L.) in a pre-Alpine river.** River Systems (formerly Large Rivers, Suppl. to Archiv für Hydrobiologie), p. n/a (accepted August 18, 2009)

Thematik: Univariate und multivariate Beschreibung von Laichplätzen der Fischarten Nase (Chondrostoma nasus, L.) und Barbe (Barbus barbus, L.) als Grundlage zur Bewertung und Verbesserung von Fischlebensräumen.

This study develops univariate utilisation- and preference indices and analyses multivariate microhabitat use of spawning nase Chondrostoma nasus (L.) and barbel Barbus barbus (L.) in the Pielach River, a pre-Alpine tributary to the Danube, Austria.During the spawning season we daily surveyed species presence, number of individuals and habitat size. Habitat features, i.e. flow velocity, water depth, shading, cover, flow protection, type of structure, substrate and embeddedness, were recorded at ten spawning grounds used by 1900 spawners within one spawning season and were compared with available habitat. Nase spawns in fast-flowing water (1 m/s) that is significantly faster than in the available habitats. In contrast, barbel constructs redds that differ in water temperature, depth, velocity and cover structure from those of nase. Multivariate analyses (PCA) showed the importance of shading and, as a consequence, the occurrence of vegetation along river banks for both fish species. This study demonstrates that efficient river restoration requires re-establishing riparian vegetation besides hydromorphological habitat improvements in order to provide adequate spawning grounds for nase and barbel.

[2] Beier U., Degerman E., Melcher A., Rogers C. & Wirlöf H. (2007): **Processes of collating a European fisheries database to meet the objectives of the European Union Water Framework Directive**. Fisheries Management and Ecology 14, 407–416

Thematik: Entwicklung einer Fisch- und Eingriffsdatenbank auf europäischer Ebene als Grundlage zur Entwicklung von Bewertungsmethoden gemäß Wasserrahmenrichtlinie.

FIsh Database of European Streams, a common database for the FAME project, was merged using existing data on electric fishing and environmental data. FIsh Database of European Stream is a relational database with eight tables. Metrics based on classification of fish species into guilds were calculated, and provided in separate tables. FIsh Database of European Stream contained information about 150 freshwater fish species, from 12 countries, 17 ecoregions, 40 main river regions, 2651 rivers and 8228 sites. Examples of data coverage and use are given. Relationships between environmental variables were illustrated using principal component analysis, which resulted in three environmental components – latitude, size and altitude. Environmental component scores were correlated with fish metrics used in the European Fish Index. Results exemplify how fish guilds reflect gradients in environmental variation. Benefits and problems concerning standardisation and data availability at the global level are discussed.

[3] Schmutz S., Melcher A., Frangez C., Haidvogl G., Beier U., Böhmer J., Breine J., Simoens I., Caiola N., de Sostoa A., Ferreira M.T., Oliveira J., Grenouillet G., Goffaux D., de Leuuw J.J., Noble R.A.A., Roset N. & Verbickas T. (2007): **Spatially based methods to assess the ecological status of riverine fish assemblages in European ecoregions**. Fisheries Management and Ecology 14, 441–452.

Thema: Entwicklung und Vergleich von auf Fischartenvergesellschaftungen basierenden Bewertungsverfahren für unterschiedliche europäische Ökoregionen.

The objective was to develop spatially based (type-specific) methods to assess the ecological status of European rivers according to the EU Water Framework Directive. Some 15 000 samples from about 8000 sites were pre-classified within a five-tiered classification system based on hydromorphological and physico-chemical pressures. The pre-classification was used to identify reference conditions and to calibrate the assessment methods.
Clustering reference sites based on relative species composition resulted in 60 fish assemblage types within 11 of the ecoregions under study. Discriminant function analyses (DFAs) were employed to identify environmental parameters characterising fish assemblage types; altitude, river slope, wetted width, mean air temperature and distance from source were the principal predictors. These environmental parameters were used to assign impacted sites with altered fish assemblage composition to the reference fish assemblage type. Metrics (fish assemblage descriptors) responding to human pressures were selected based on correlation and DFAs. Assessment methods were developed for 43 fish assemblage types. Metrics based on individual sentinel species were more often used in type-specific methods than metrics related to reproduction, habitat and feeding. Metrics based on long-distance migrants and potamodromous species were more sensitive to human pressures than overall composition metrics, e.g. total number of species. Only some of the tested metrics showed pressure-specific responses, i.e. reacted to one type of pressure but not to others. Insectivorous, intolerant and lithophilic species exclusively responded (decreased) to chemical and hydromorphological pressures in 14–19%. Omnivorous species was the only metric type that showed a consistent reaction (increase) to continuum disruptions in 25% of the cases. Accuracy of methods based on cross-validation with pre-classification varied between 47% and 98% (mean 81%) when contrasting calibration data set (class 1 and 2) with degraded sites (class 3, 4 and 5).

[4] Melcher A., Schmutz S., Haidvogl G. & Moder K. (2007): **Spatially based methods to assess the ecological status of European fish assemblage types**. Fisheries Management and Ecology 14, 453–463.

Thema: Identifikation und Beschreibung von Fischartenvergesellschaftungen in Europa, sowie die Entwicklung und der Vergleich von Bewertungsmethoden auf gesamteuropäischer Ebene.

A spatially based, river type-specific approach was used to develop an ecological assessment method for European rivers based on existing sampling data. The methodology comprised two main steps: (1) description of a river and fish assemblage typology based on minimally or slightly impacted sites and (2) analyses of impacted conditions for each type. Hierarchical cluster analysis of fish species assemblages identified 15 homogeneous groups in 11 European ecoregions. Discriminant analyses, based on abiotic characteristics, were used to predict fish types at impacted sites. The latter encompassed both regional (geographic position in Europe) and local factors (longitudinal zonation) influencing the distribution of riverine fishes. To assess ecological status, the responses of more than 400 metrics (species composition, abundance and age–length structure) to human pressures were tested for each river type separately. A maximum of 10 metrics per river type was selected using discriminant analysis. The density of intolerant species and feeding guilds had the highest capacity to predict the intensity of perturbation.

[5] Pont D., Hugueny B., Beier U., Goffaux D., Melcher A., Noble R., Rogers C., Roset N. & Schmutz S. (2006): **Assessing river biotic condition at a continental scale: a European approach using functional metrics and fish assemblages**. Journal of Applied Ecology 43, 70–80.

Thema: Entwicklung einer europäischen Bewertungsmethode gemäß EU Wasserrahmenrichtlinie für Fische - dem EFI (European Fish Index).

1. The need for sensitive biological measures of aquatic ecosystem integrity applicable at large spatial scales has been highlighted by the implementation of the European Water Framework Directive. Using fish communities as indicators of habitat quality in rivers, we developed a multi-metric index to test our capacity to (i) correctly model a variety of metrics based on assemblage structure and functions, and (ii) discriminate between the effects of natural vs. human-induced environmental variability at a continental scale.
2. Information was collected for 5252 sites distributed among 1843 European rivers. Data included variables on fish assemblage structure, local environmental variables, sampling strategy and a river basin classification based on native fish fauna similarities accounting for regional effects on local assemblage structure. Fifty-eight metrics reflecting different aspects of fish assemblage structure and function were selected from the available literature and tested for their potential to indicate habitat degradation.
3. To quantify possible deviation from a 'reference condition' for any given site, we first established and validated statistical models describing metric responses to natural environmental variability in the absence of any significant human disturbance. We considered that the residual distributions of these models described the response range of each metric, whatever the natural environmental variability. After testing the sensitivity of these residuals to a gradient of human disturbance, we finally selected 10 metrics that were combined to obtain a European fish assemblage index. We demonstrated that (i) when considering only minimally disturbed sites the index remains invariant, regardless of environmental variability, and (ii) the index shows a significant negative linear response to a gradient of human disturbance.
4. Synthesis and applications. In this reference condition modelling approach, by including a more complete description of environmental variability at both local and regional scales it was possible to develop a novel fish biotic index transferable between catchments at the European scale. The use of functional metrics based on biological attributes of species instead of metrics based on species themselves reduced the index sensitivity to the variability of fish fauna across different biogeographical areas.

[6] Degerman E., Beier U., Breine J., Melcher A., Quataert P., Rogers C., Roset N. & Simoens I. (2007): **Classification and assessment of degradation in European running waters**. Fisheries Management and Ecology 14, 417–426.

Thema: Betrachtung von hydromorphologischen und chemischen Beeinträchtigungen in europäischen Flüssen.

A pan-European, classification of the extent of environmental degradation from chemical, physical and biological pressures on fish communities as a precursor to assess the ecological status of running waters based on fish is proposed. Twenty-four potential pressures acting on fish communities at three different spatial scales (river basin, segment and site) were identified and class boundaries for high, good, moderate, poor and bad status, based on existing data and/or expert judgement, were defined. Four pressures (hydrological regime, morphological conditions, toxic or acid conditions, nutrients and organic load) were found to describe the majority of degradation at a specific site and these were combined into a single pressure variable to describe impact at each location. Principal Component Analysis showed that the four variables were correlated with other physical and chemical variables not included in the combined pressure variable. However, biological pressures, e.g. introduction of fish, and longitudinal connectivity were not well correlated, suggesting that two dimensions of human impact on stream fish were poorly accounted for. Low-resolution Geographical Information Systems (GIS) data (1 km grid) on land use and population density correlated well with the four chosen pressures, suggesting it is possible to use standardised GIS data to aid pre-classification of stream degradation.

[7] Schinegger R., Melcher A., Trautwein C. & S. Schmutz (2009): **Detecting patterns and relationships of human pressures in European rivers**. International Association of Hydraulic Engineering and Research, 33rd IAHR Congress: Water Engineering for a Sustainable Environment; 4702 – 4709. ISBN: 978-94-90365-0

Thema: Betrachtung und Analyse von unterschiedlichsten anthropogenen Beeinträchtigungen sowie deren Kombinationen in europäischen Flüssen.

Most European rivers are affected by different types of human pressures that may impair fish populations. We analysed 15 pressure variables of 4 different pressure groups, i.e. hydrology, morphology, water quality and connectivity to detect spatial patterns, relationships and interactions between pressures and natural environment at the European scale. Based on literature, national databases and expert knowledge important pressures were identified and collected within the EU-project EFI+ in 14 countries at about 10 000 fish-sampling sites in Europe. In 90% of the catchments analysed fish migration was interrupted by barriers. We used PCA and correlation analysis to identify key pressures and to eliminate redundant pressures at local and river segment scale. Thirteen variables were found to describe the majority of human degradation at a specific site. To aggregate into pressure type specific indices we first harmonized the variables along a gradient from 1-5, i.e. from nearly undisturbed to strongly impacted sites. Further, we calculated the mean of values > class 2 only, to avoid that values <=2 compensate values >2, i.e. to better indicate degradation. Pressure analysis showed that 24% of sites are affected by single, 22% by double 19% by triple and 12% by four pressure groups. Only 23% of sites are less affected, i.e. class <=2. In terms of pressure types, analysed sites showed alterations in 55% for water quality pressures, 40% for hydrology, 37% for morphology and 34% for connectivity (river segment). In 45% of the cases water quality problems are also associated with other pressures. The results clearly show that European rivers are multi-impacted. Therefore, only restoration strategies simultaneously considering all important types of pressures will guarantee the achievement of the good ecological status or potential sensu EU Water Framework Directive.

[8] Matulla C., Schmutz S., Melcher A., Gerersdorfer T. & Haas P. (2007):
Assessing the impact of a downscaled climate change simulation on the fish fauna in an Inner-Alpine River. INT J BIOMETEOROL, 52, 127-137; ISSN 0020-7128

Thema: Auswirkungen der klimabedingten Erwärmung werden anhand der Verschiebung von Fischregionen an der Mur (Ö) erklärt.

This study assesses the impact of a changing climate on fish fauna by comparing the past mean state of fish assemblage to a possible future mean state. It is based on (1) local scale observations along an Inner-Alpine river called Mur, (2) an IPCC emission scenario (IS92a), implemented by atmosphere-ocean global circulation model (AOGCM) ECHAM4/OPYC3, and (3) a model-chain that links climate research to hydrobiology. The Mur River is still in a near-natural condition and water temperature in summer is the most important aquatic ecological constraint for fish distribution. The methodological strategy is (1) those downscaled air temperature and precipitation scenarios for the first half of the twenty-first century, (2) to establish a model that simulates water temperature by means of air temperature and flow rate in order to generate water temperature scenarios, and (3) to evaluate the impact on fish communities using an ecological model that is driven by water temperature. This methodology links the response of fish fauna to an IPCC emission scenario and is to our knowledge an unprecedented approach. The downscaled IS92a scenarios show increased mean air temperatures during the whole year and increased precipitation totals during summer, but reduced totals for the rest of the annual cycle. These changes result in scenarios of increased water temperatures, an altered annual cycle of flow rate, and, in turn, a 70 m displacement in elevation of fish communities towards the river's head. This would enhance stress on species that rely on low water temperatures and coerce cyprinid species into advancing against retreating salmonids. Hyporhithral river sectors would turn into Epipotamal sectors. Grayling (Thymallus thymallus) and Danube salmon (Hucho hucho), presently characteristic for the Mur River, would be superceded by other species. Native brown trout (Salmo trutta), already now under pressure of competition, may be at risk of losing its habitat in favour of invaders like the exotic rainbow trout (Oncorhynchus mykiss), which are better adapted to higher water temperatures. Projected changes in fish communities suggest an adverse influence on salmonid sport fishing and a loss in its high economic value.

ZUSAMMENFASSUNG UND DISKUSSION DER PUBLIKATIONEN

Ausgangsbasis vorliegender Dissertation ist die Publikation [1], welche grundsätzlich die Problematik der Quantifizierung von Lebensraumansprüchen behandelt. An weniger beeinträchtigten Flüssen, wie z.B. der Pielach in Niederösterreich, können zum Teil noch intakte Lebensräume der Fischarten Nase *Chondrostoma nasus* und Barbe *Barbus barbus*, durch abiotische Parameter beschrieben werden. Dabei werden auf Ebene von Mikrohabitaten Faktoren wie die Wassertiefe, die Fließgeschwindigkeit oder die Größe des Substrates gemessen. Zusätzlich werden auch Strukturparameter, Beschattung, Sicht- und Strömungsschutz für Fische erhoben.

Mit Hilfe von Nutzungs- und Präferenzkurven lassen sich Laichhabitate von Nase und Barbe univariat beschreiben. Multivariate Habitatmodelle (PCA Hauptkomponentenanlyse), welche in der Analyse mehrere unabhängige Einflussgrößen berücksichtigen, erlauben den Schluss, dass neben einer hydromorphologischen Komponente auch eine strukturbezogene (Ufervegetation) besteht. Die gewonnenen Erkenntnisse dienen als Grundlage zur Bewertung und Verbesserung von Fischlebensräumen. In Folge soll dadurch auch die ökologische Funktionsfähigkeit von Fließgewässern hergestellt werden.

Der Begriff der ökologischen Funktionsfähigkeit wurde bereits im Jahre 1985 im österreichischen Wasserrecht (§ 105 (1) lit. M.) verankert und verlangt eine ökologisch orientierte ganzheitliche Betrachtungsweise von Fließgewässern. Definitionsgemäß wird die ökologische Funktionsfähigkeit als die Fähigkeit zur Aufrechterhaltung des Wirkungsgefüges zwischen dem Gewässer und seinem Umland gegebenen Lebensraum und seiner organismischen Besiedelung entsprechend der natürlichen Ausprägung des betreffenden Gewässertyps (ÖMORM M 6232) gesehen.

Mit dem Inkrafttreten der EU Wasserrahmenrichtlinie (WRRL) im Jahr 2000 wurden die nationalen Ziele zur Verbesserung der ökologischen Funktionsfähigkeit auf Europa ausgeweitet. Die EU-Mitgliedsstaaten müssen ihre Gewässer schützen und verbessern, um bis 2015 auf einer 5-stufigen Skala einen „guten Gewässerzustand“ (Stufe 2) zu erreichen. Als Referenz wird der vom Menschen nicht veränderte Naturzustand der Gewässer verwendet (Stufe 1). Die Bewertung erfolgt anhand von biologischen Indikatorgruppen. Neben dem Phytoplankton, den Makrophyten und dem Phytobenthos und der benthischen wirbellosen Fauna ist auch die Fischfauna Indikator für den Gewässerzustand.

Zur Realisierung dieser Gewässerbewertung hat sich das von der EU finanzierte Forschungsprojekt FAME „Development, Evaluation and Implementation of a Standardised Fish-based Assessment Method for the Ecological Status of European Rivers. A Contribution to the Water Framework Directive" in den Jahren 2002 bis 2004 zum Ziel gesetzt, eine fischbezogene Bewertungsmethode für den ökologischen Zustand von europäischen Flüssen zu entwickeln. Das Projekt baut auf bereits vorhandenen Fischdaten aus 16 von insgesamt 25 europäischen „Ökoregionen" auf. Mit Daten von ca. 17.000 Probestellen aus etwa 5000 Fließgewässern werden die Referenzzustände und die Abweichungen im Sinne der Wasserrahmenrichtlinie für unterschiedliche Flusstypen modelliert. Im FAME Projektteam waren 23 Partner aus 11 europäischen Ländern vertreten.

Die Publikationen [2, 3, 4, 5, 6] entstanden im Rahmen dieses FAME Projektes (Schmutz et al. 2007).

Eine wesentliche Grundlage zur Bewertung von Gewässern bildet die Entwicklung einer Datenbank auf europäischer Ebene, wie in der Publikation [2] beschrieben. Die Fish Database of European Streams (FIDES), enthält neben Fisch- auch Eingriffsdaten sowie ausgewählte Umweltparameter. Sie besitzt einen hierarchischen Aufbau und ist in acht Tabellen gegliedert.

Die kleinste Einheit ist dabei eine einzelne Probenstelle. Neben der Anzahl gefangener Fische enthält sie auch Informationen zur Beprobungsmethode und der Größe der Probenstelle. Diese Angaben ermöglichen vorher definierte Metrics zu berechnen. Die in der WRRL geforderten Metrics zur Bewertung des ökologischen Zustandes beinhalten die Artenzusammensetzung (incl. Ernährungsgilden, Reproduktionsgilden und Gilden zur Lebensweise), die Abundanz und den Altersaufbau (Roset et al. 2007). Basis zur Berechnung von Metrics bildet eine Einstufung aller, zumindest in FIDES, vorkommender Fischarten und deren Einteilung in Gilden (Noble et al. 2007).

Mittels deskriptiver Statistik und der Hauptkomponentenanlyse (PCA) werden die Datensätze auf ihre Repräsentativität überprüft und die natürliche Variabilität von Fischartenvergesellschaftungen erklärt. Dadurch können Vorteile und Probleme betreffend der Standardisierung und Datenverfügbarkeit auf internationaler Ebene diskutiert werden.

Die klare und übersichtliche Struktur von FIDES ermöglicht auch ihre Verwendung auf nationaler Ebene und ebenso für fischökologisch orientierte Projekte im Allgemeinen (z.B. STAR, EFI+, WISER).

FIDES samt den berechneten Metrics bildet die Basis für die folgenden Publikationen [3, 4, 5, 6]. Während sich die Publikationen [3], [4] und [5] mit der Entwicklung von Bewertungsmethoden auf Basis der Qualitätskomponente Fisch beschäftigen, ist es Ziel der Publikation [6], anthropogene Eingriffe an Europäischen Fließgewässern zu analysieren.

Erste Grundlagen für die angesprochenen Bewertungsmethoden wurden bereits von Karr (1981) für den Index of Biotic Integrity (IBI) zusammengefasst. Dieser beruht (1) auf einer Beschreibung der Fischartengemeinschaft durch einzelne Metrics, (2) auf der Reaktion von Fischartengemeinschaften oder eben Metrics für anthropogene Eingriffe und (3) einem Expertensystem zur Evaluierung dieser.

In vorliegender Dissertation werden grundsätzlich zwei Arten von Bewertungsmethoden unterschieden, nämlich die „spatially based“ Methode mit einem Typspezifischen Ansatz (Publikationen [3, 4]) und einem multimetrischen Index (Publikation [5]).

Beiden Arten liegen folgende Überlegungen zugrunde:

- Die Definition von Referenzzuständen (Reference conditions)
- Entwicklung einer Gewässertypologie (River typology) basierend auf dem jeweiligen natürlichen Fischartenvorkommen
- Vorhersage des jeweiligen Gewässertyps mithilfe unabhängiger abiotischer Parameter (z.B. Seehöhe, Gefälle, Entfernung zur Quelle etc.)
- Die Voreinstufung von anthropogenen Belastungen
- Die Zuordnung von Fischarten in einzelne Gilden (Species classification)
- Die Definition und Berechnung von Metrics
- Die Berücksichtigung von standardisierten Probennahmen (Sampling standard)

In der Publikation [3] wurden „spatially based“ Methoden für jede Ökoregionen einzeln entwickelt. Dabei wurde auf Basis der relativen Häufigkeit einzelner Fischarten vornehmlich mittels Clusteranlyse Gewässertypen definiert. Es wurden 60 verschiedene Fischartengemeinschaften in 11 Ökoregionen gefunden und in weiterer Folge für jede einzelne dieser Gemeinschaften eine eigene Bewertungsmethode entwickelt. Trotz des enormen Datenvolumens, welches in FIDES enthalten war, lagen zumeist zu wenige Datensätze, speziell Referenz- oder Kalibrierungsstellen, vor, um für alle Typen standardisierte Methoden zu entwickeln und zu validieren.

Aufgrund dieser Erkenntnisse und der Überlegung, dass sich einige der 60 oben genannten Gewässertypen und deren Bewertungsmethoden in verschiedenen Ökoregion sehr ähneln würden, wird die „spatially based“ Methode in der Publikation [4] mit Hilfe des gesamteuropäischen Datensatzes weiterentwickelt.

Dabei kann im ersten Schritt die Anzahl an relevanten Fischartengemeinschaften von 60 auf 15 reduziert werden. Mit Hilfe der abiotischen Parameter Seehöhe, Gefälle, Entfernung zur Quelle, Lufttemperatur, Gewässerbreite und geographische Lage können diese 15 EFT (European Fish Types) unterschieden werden. Dadurch kann mit den Koeffizienten von Diskriminanzanalysen, welche in die FAME Software (http://fame.boku.ac.at/downloads.htm) integriert wurden, bei Kenntnis dieser Parameter für jeden beliebigen Gewässerabschnitt in Europa die Fischartengemeinschaft vorhergesagt werden.

Auf Basis dieser Typen und nach Auswahl geeigneter signifikanter Metrics wurde schlussendlich die „spatially based“ Methode für Europa (SBM EU) entwickelt. Diese Methode beinhaltet 15 Diskriminanzmodelle (für jeden EFT separat), um den ökologischen Zustand prognostizieren zu können.

Im Gegensatz dazu beruht der eigentliche spätere EFI (European Fish Index, Pont et al. 2007) auf einer multimetrischen Methode. Diese ist in der Publikation [5] beschrieben. Dabei wird in lediglich einem umfassenden Modellansatz, welcher auch die oben erwähnten abiotischen und biotischen Parameter (natürliche Variabilität) berücksichtigt, mit Hilfe von multiplen Regressionsmodellen und deren Abweichungen vom Referenzzustand (Residuen) der ökologische Gewässerzustand berechnet. Insgesamt wurden 58 potentielle Metrics aus mehr als 300 verschiedenen getestet, wobei lediglich 10 Metrics in der endgültigen Methode berücksichtigt wurden. Alle angeführten Bewertungsmethoden stoßen vor allem im mediterranen Raum, in ausschließlich hydromorphologisch belasteten und in sehr großen Flüssen an die Grenzen ihrer Aussagekraft. Alle im Rahmen des FAME Projektes entwickelten Methoden sind von Quataert et al. (2007) gegenübergestellt und miteinander verglichen worden.

Eine Übertragbarkeit beider europaweiten Methoden („spatially based“ und „multi metrisch“) auf andere räumliche Betrachtungsebenen ist grundsätzlich möglich, sofern genügend Kalibrations- oder Referenzstellen vorhanden sind. Außerdem ist die Betrachtung und Differenzierung von Gewässertypen (vgl. EFT) für alle Analysen des Wirkungsgefüges Fisch – Belastung unerlässlich. Dies haben auch Erfahrungen aus neueren Projekten wie MIRR (Schmutz et al. 2008) und EFI+ gezeigt. Schließlich wurden auch im FAME Folgeprojekt EFI+ zur Verbesserung des Europäischen Fischindex die EFT zur Abgrenzung von Salmonidengewässern verwendet und deren Prognosemodell im Softwarepaket implementiert (http://efi-plus.boku.ac.at/software/).

Diese Software stellt die Basis für die zukünftige Bewertung von Flüssen gemäß WRRL dar und ist frei zugänglich. Neben ihrer Bedeutung für die Interkalibrierung im Rahmen der WRRL (Van de Bund 2004) wird sie in Ländern eingesetzt, denen keine eigens auf nationaler Ebene entwickelte Bewertungsmethode vorliegt. Mit der von der WRRL vorgesehenen Interkalibrierung soll eine Vergleichbarkeit der Ergebnisse der in den Mitgliedstaaten zu implementierenden biologischen Bewertungsverfahren gewährleistet werden.

Gerade wenn von Grenzen und Möglichkeiten von Bewertungsmethoden gesprochen wird, darf der Aspekt inwieweit diese belastungsspezifisch reagieren nicht vergessen werden. Die folgenden 3 Publikationen beschäftigen sich speziell mit der Analyse anthropogener Belastungen.

Die Publikation [6] setzt sich im Rahmen des FAME Projektes mit der Analyse (Deskriptive Statistik und Faktoranalysen) von hydromorphologischen und chemischen Beeinträchtigungen in europäischen Flüssen auseinander. Eine Fortsetzung dieser Betrachtungen, vor allem in Hinblick auf Mehrfachbelastungen und mögliche Kombinationen von Beeinträchtigungen wurde vor allem im Projekt EFI+ vorangetrieben. Aus diesem Projekt entstammt die Publikation [7]. Der dabei entwickelte Belastungsindex soll auf Basis hydromorphologischer (incl. Kontinuum) und chemischer Belastungen den Grad der Beeinträchtigung genauer abgrenzen. Dabei dient er als Grundlage für zukünftige Wirkungsmodelle, welche die Reaktion von Fischen auf unterschiedliche anthropogene Beeinträchtigungen und deren Grad beschreibt.

Eine zusätzliche Belastung für Gewässer und deren Organismen die bisher in Zusammenhang mit Bewertung und Restauration kaum berücksichtigt wurde, ist die Klimaerwärmung. Die Publikation [8] hat deren Auswirkungen auf die Fischfauna zum Inhalt. Wie auch jüngste Studien (Melcher et al. 2009) und unter anderem auch das laufende EU Projekt WISER zeigen, wird in Zukunft die Klimaerwärmung für die Bewertung des ökologischen Zustandes enorm an Bedeutung gewinnen und eine besondere Stellung für weitere zukünftige Forschungsaufgaben einnehmen müssen.

Die Erkenntnisse dieser Arbeit sollen unter anderem auch ermöglichen, dass in Gebieten außerhalb Europas, wie z.B. in Afrika, der Zustand von Fließgewässern bewertet und gegebenenfalls verbessert werden kann.

LITERATUR

Amtsblatt der Europäischen Gemeinschaften (2000): Ausgabe L327 vom 22.12.2000 (43. Jahrgang, ISSN 0376-9453) © Europäische Union

Jungwirth, M., Haidvogl, G., Moog, O., Muhar, S., Schmutz, S. (2003): Angewandte Fischökologie an Fließgewässern. 552; Facultas Universitätsverlag,Wien; ISBN 3-8252-2113-X

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

Melcher A., J. Huber, F. Pletterbauer, C. Walder und S. Schmutz (2009): Wärmelastplan Traun- und Agersystem. Bewertung des fischökologischen Zustands der Traun und Ager unter besonderer Berücksichtigung thermischer Belastungen . Amt der Oberösterreichischen Landesregierung. Abteilung Wasserwirtschaft Wasserwirtschaftliches Planungsorgan. 4021 Linz, Kärntnerstraße 12 , 187

Melcher A., J. Huber, S. Schmutz (2009): Climate Change Impacts on Fish in Austrian Rivers. In: Pröbstl, U. (Eds.), Book of Abstracts, 15th International Symposium on Society and Resource Management, Vienna. [ISSRM 2009 - 15th International Symposium on Society and Resource Management. Meet old and new worlds in Research, Planning and Management, Vienna, July 5-8, 2009]

Noble R.A.A., Cowx I.G., Goffaux D. & Kestemont P. (2007): Assessing the health of European rivers using functional ecological guilds of fish communities: standardising species classification and approaches to metric selection. Fisheries Management and Ecology 14, 381–392.

Pont D., Hugueny B. & Rogers C. (2007) Development of a fish-based index for the assessment of river health in Europe: the European Fish Index. Fisheries Management and Ecology 14, 427–439.

Quataert P., Breine J. & Simoens I. (2007): Evaluation of the European Fish Index: false-positive and false-negative error rate to detect disturbance and consistency with alternative fish indices. Fisheries Management and Ecology 14, 465–472.

Roset N., Grenouillet G., Goffaux D., Pont D. & Kestemont P. (2007): A review of existing fish assemblage indicators and methodologies. Fisheries Management and Ecology 14, 393–405.

Schmutz S., Cowx I.G., Haidvogl G. & Pont D. (2007): Fish-based methods for assessing European running waters: a synthesis. Fisheries Management and Ecology 14, 369–380.

Schmutz S., Kaufmann M., Vogel B., Jungwirth M. & Muhar S. (2000): A multi-level concept for fish-based, river-type-specific assessment of ecological integrity. In: M. Jungwirth, S. Muhar & S. Schmutz (eds) Assessing the Ecological Integrity of Running Waters. Dordrecht: Kluwer Academic Publishers, pp. 279–289.

Schmutz, S., M. Kaufmann, B. Vogel, M. Jungwirth & S. Muhar (1999): Methodische Grundlagen und Beispiele für die Bewertung der fischökologischen Funktionsfähigkeit österreichischer Fließgewässer. Studie im Auftrag des BMLF, Wien.

Schmutz, S., A. Melcher, S. Muhar, A. Zitek, M. Poppe, C. Trautwein, M. Jungwirth (2008): MIRR – Model-based Instrument for River Restoration. Entwicklung eines strategischen Instruments zur integrativen Bewertung ökologischer Restaurationsmaßnahmen an Fließgewässern. Österreichische Wasser- und Abfallwirtschaft, 80, 95-103

Van de Bund W. (2004): Guidance on the Intercalibration Process [version 4.1]. ECOSTAT WG 2.A. ISPRA

PROJEKTE IM INTERNET

EFI+ Improvement and Spatial extension of the European Fish Index

A specific targeted research project supported by the European Commission under FP 6 contributing to the implementation of task "Ecological Status Assessment - filling the gaps", Priority "Integrating and Strengthening the European Research Area - Scientific Support to Policies", Task 4 Contract Number 044096

http://efi-plus.boku.ac.at

FAME Development, Evaluation and Implementation of a standardised Fish-based Assessment Method for the Ecological Status of European Rivers

A project under the 5th Framework Programme Energy, Environment and Sustainable Development, Key Action 1: Sustainable Management and Quality of Water) Contract No: EVK1 -CT-2001-00094

http://fame.boku.ac.at

MIRR Model-based Instrument for River Restoration. Development of a strategic instrument for integrated assessment of ecological restoration measures in running waters

Entwicklung eines strategischen Instruments zur integrativen Bewertung ökologischer Restaurationsmaßnahmen an Fließgewässern. Im Auftrag des BMLFUW gemeinsam mit dem Land NÖ. Das Projekt wurde mit BMLFUW, Sektion Wasser, Ref. VII 1 b – Gewässerökologie und VII-5 – Schutzwasserwirtschaft akkordiert.

http://mirr.boku.ac.at

STAR Standardisation of River Classifications

Framework method for calibrating different biological survey results against ecological quality classifications to be developed for the Water Framework Directive. The Star project is a research project supported by the European Commission under the Fifth Framework Programme and contributing to the implementation of the Key Action "Sustainable Management and Quality of Water" within the Energy, Environment and Sustainable Development Programme. Contract No: EVK1-CT 2001-00089

http://www.eu-star.at

WISER Water bodies in Europe: integrative systems to assess ecological status and recovery

The WISER project is a research project supported by the European Commission under the Seventh Framework Programme. Theme 6 (Environment including Climate Change), Contract No.: 226273

http://www.wiser.eu

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The importance of structural riverine features in spawning habitat of nase *Chondrostoma nasus* (L.) and barbel *Barbus barbus* (L.) in a pre-Alpine river

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This study develops univariate utilisation- and preference indices and analyses multivariate microhabitat use of spawning nase *Chondrostoma nasus* (L.) and barbel *Barbus barbus* (L.) in the Pielach River, a pre-Alpine tributary to the Danube, Austria. During the spawning season we daily surveyed species presence, number of individuals and habitat size. Habitat features, i.e. flow velocity, water depth, shading, cover, flow protection, type of structure, substrate and embeddedness, were recorded at ten spawning grounds used by 1900 spawners within one spawning season and were compared with available habitat. Nase spawns in fast-flowing water (1 m/s) that is significantly faster than in the available habitats. In contrast, barbel constructs redds that differ in water temperature, depth, velocity and cover structure from those of nase. Multivariate analyses (PCA) showed the importance of shading and, as a consequence, the occurrence of vegetation along river banks for both fish species. This study demonstrates that efficient river restoration requires re-establishing riparian vegetation besides hydromorphological habitat improvements in order to provide adequate spawning grounds for nase and barbel.

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Introduction

Nase *Chondrostoma nasus* (L.) and barbel *Barbus barbus* (L.) are the two dominating and co-existing cyprinid species in the barbel zone of Austrian rivers. Knowledge of their spawning habitat requirements in rivers is important to understand their population biology. Moreover, knowledge of their microhabitat requirements would be valuable when planning management schemes to mitigate the effects of habitat alteration caused by channelisation, continuum disruption and impoundments.

A lot of studies on microhabitat use focused on salmonids (e.g. Northcote 1984, Shirvell 1989, Wollebaek et al. 2008, Moir and Pasternack 2009) while the knowledge on spawning habitat of riverine cyprinids is scarce and only few studies were done in the Danube catchment (Kekeis et al. 1996, Rakowitz et al. 2008).

Spawning behaviour of nase were studied by field surveys using electrofishing and radio telemetry in Belgium (Ovidio and Phillipart 2008), Switzerland (Zbinden and Maier 1996, Huber and Kirchhofer 1998 and 2001) and Czech Republic (e.g. Lelek and Penaz 1963, Penaz 1996). Spawning habitat preferences of barbel are described exclusively by Baras (1992, 1994 and 1997) based on studies in Belgium. Nase and barbel are classified as lithophilic spawners (Balon 1975b), however, there is no detailed information on the differences of spawning habitats available.

A number of alternative physical habitat models have been developed world wide, introducing more effective sampling methods and multidimensional hydraulics as well as more user-friendly software solutions (Parasiewicz and Dunbar, 2001). Significant effort was also invested in better description of ecological interactions by application of multivariate-statistical or mechanistic approaches opening the way to optimise the multi-factorial character of influences on the aquatic biota (Santos and Ferreira 2008, Wollebaek et al. 2008). In addition, a wider range of numeric and categorical physical attributes can be used for model construction (e.g. velocity and cover structure) increasing the understanding of multi-dimensionality of habitat use (Ahmadi-Nedushan et al. 2006). Since it is expected that aquatic species are stimulated rather by cumulative effects of many habitat variables than by individual effects methods for quantification of integrative effects have to be elaborated.

The objectives of this study were to (1) determine the microhabitat utilization and preferences of nase and barbel, (2) to identify habitat features as basic requirements for spawning, and (3) to investigate explanatory power of physical variables to characterise spawning site microhabitats.

Study area

The study was conducted in the pre-Alpine River Pielach, a medium sized tributary of the River Danube. The area is situated about 100 km west of Vienna below the village of Grafendorf in Lower Austria (15°30´38´´ E and 48°14´24´´ N; WGS 84) (Figure 1). The distance to the mouth of the Danube into the Black Sea amounts 2034 kilometres.

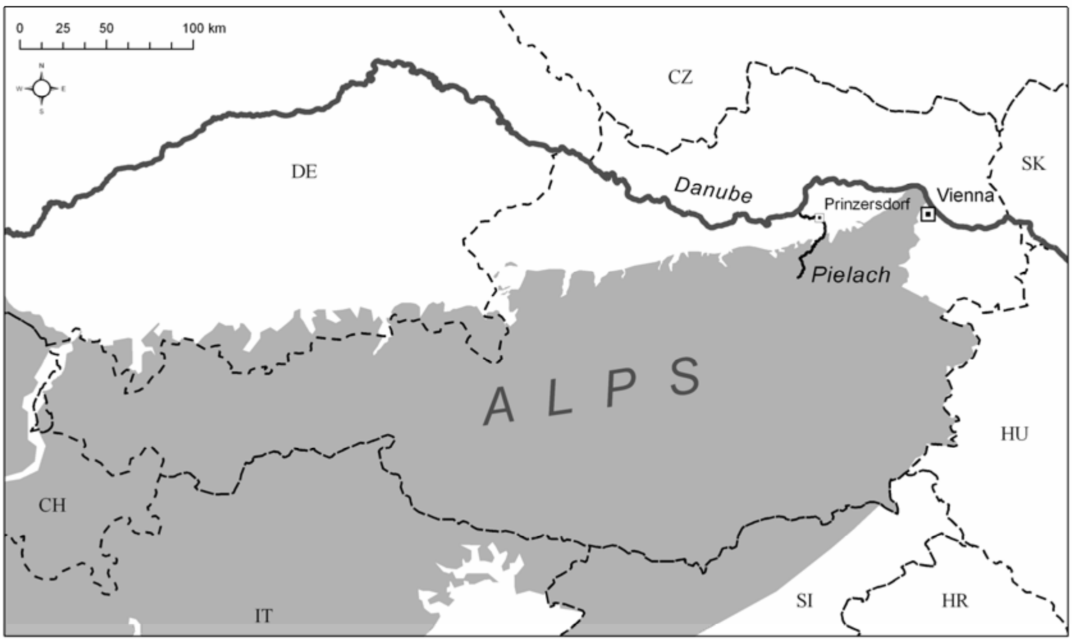

Figure 1: Study area located in Lower Austrian pre-Alpine area.

The source of the River Pielach is located at an elevation of about 1000 m above see level and the river flows 68 km northwards down to the Danube (208 m above see level) reaching stream order 4. The catchment size is 591 km² and the mean annual precipitation is 875 mm. Mean annual low flow is equal to 2 m^3/s, mean annual flow is 6.5m^3/s and mean annual high flow is about 240 m^3/s. Consequently the river is characterized by flashy flows and a relatively high gradient (>0.3%). In its lower part the Pielach changes its character from meandering-braided to meandering. Here, it passes through small villages and agricultural land. The continuum in the Pielach is disrupted by weirs built for water abstraction and engineering measures, but it is possible for fish to enter the first two kilometres from the Danube. Nevertheless the Pielach has retained some of its natural morphological characteristics such as meandering sections, side arms, dynamic gravel bars, large woody debris, small oxbows, inundation areas and floodplain forests (Zitek et al. 2008).

The studied river section is located in the lower Pielach between km 0 and 35 with a mean slope of 0.15 % (typical of the barbel zone; Huet, 1949) and a width of 15–25 m. Water temperature and mean flow was monitored continuously at upstream gauging station (river km 30) and additionally with a thermograph close to the spawning area. Daily mean spring water temperature (April–June) ranged from 5.5 to 18.5°C. Mean daily flow varied from 4.7 to 42.6 m^3/s.

The fish community was dominated by nase (20 %) and barbel (20 %). Other abundant species out of 29 were *Alburnoides bipunctatus* (Bloch), *Barbatula barbatula* (L.), *Cottus gobio* (L.), *Gobio gobio* (L.), *Hucho hucho* (L.), *Leuciscus cephalus* (L.), *Leuciscus leuciscus* (L.), *Phoxinus phoxinus* (L.) and *Salmo trutta* (L.) (Zitek et al. 2004).

Methods

The studied river section was surveyed daily from March to June until the first spawners were observed at the spawning sites. The substrate at the spawning redds was checked for eggs to confirm that spawning had indeed started.

Both nase and barbel spawned in shoals on shallow gravel bars easily to identify from the river bank. Spawning took place other several days and superimposition of spawning redds was very common. Therefore, a grid of equally spaced points was laid over the spawning area (grid size 1 m^2). Due to the very clear water and shallow habitats it was possible to count spawning individual by visual observations (see Figure 2 and 3).The number of measured points approximately equalled the number of observed spawning fish (nase n=1250, barbel n=610).

Additionally, we sampled representative sites of different morphological characteristics within the 35 km long study area to describe the available habitat in the River Pielach. We made point measurements at 71 transects with a distance of 2 m within transects resulting in 582 points.

Figure 2: Spawning site full of nase, located in the Pielach River 1.3 km upstream the mouth into the Danube. Dark spots indicate spawning fish over coarse substrate, on the left site middle “splashing” nases actively spawning.

Figure 3: Spawning site of barbel, located in the Pielach River 1 km upstream the mouth into the Danube.

We measured the following habitat variables: water depth with a scale, flow velocity (v) at the bottom (5 cm above bottom), in 40 % of water depth and at the water surface using inductive flow meter (Flo-Mate®), distance from nearest bank, shading (yes, no), cover (yes, no), cover type (boulder, broken water surface, overhanging riparian vegetation, submerged vegetation, rip-rap, undercut bank), effective visual cover for the fish (yes, no), flow protection (yes, no), dominating substrate (pelal, psammal <2, akal 2-20, microlithal 20-63, mesolithal 63-200, macrolithal 200-400, megalithal >400 mm) and substrate embeddedness (no, means loose and yes, means embedded) estimated by kicking with the rubber boots against the bottom.

The mean flow velocity (v-mean) was calculated according to the two-point (2P) method formula (Bretschneider et al. 1993) as follows:

[1] $$v\text{-}mean = 0.31 * \cdot v\text{-}40 + 0.634\ v\text{-}surface$$

where *v-40* is flow velocity at 40 % of water depth and *v-surface* is flow velocity at the water surface.

Statistical analyses were conducted with SPSS 15.0®. Categorical variable cover type was transformed into binary variable and coded as 0 and 1. Ordinal variable substrate was coded from 1 (pelal) to 7 (megalithal) resulting in a total of 15 variables tested. Statistical significance levels were set at <=0.05 (‘‘strong significance’’) and <=0.1 (‘‘weak significance’’). Non-normality in data and variation in standard deviations for many variables dictated the use of nonparametric statistics in comparative tests (Sokal and Rohlf 1981). Mann Whitney *U*-Test was used to compare habitat use of fish species. Correlation matrices (Pearson´s and Spearman correlation coefficient, r) were used to investigate relations among the measured microhabitat variables. Available habitat data and utilisation data from spawning habitats were tested separately. An additional method was used: measures of effect size in ANOVA, which are measures of the degree of association between and effect (e.g., a main effect, an interaction) and the dependent variable. They can be thought of as the correlation between an effect and the dependent variable. If the value of the measure of association is squared it can be interpreted as the proportion of variance in the dependent variable that is attributable to each effect. One of the commonly used measures of effect size is Eta squared (η^2) for showing the importance of single variables in discriminating spawning habitats of nase and barbel (Kirk 1982, Tabachnick and Fidell 1989). Furthermore, principal component analyses (PCA) were applied using rotation method, Varimax with Kaiser Normalisation.

Finally, available habitat, suitability and preference curves were developed for each microhabitat variable using frequency-of-use graphs (FUG, Raleigh et al. 1986) as normalized probability density function ranging from 0 to 1.

[2] $$FUG_i = f_i / f_{[max]}$$

where: f_i is class frequency and $f_{[max]}$ is maximium class frequency.

For preference curves we used the Ivlev index (Ivlev 1961):

[3] $$Preference = U / A$$

where: U is class frequency of habitat used and A class frequency of habitat available.

Results

Migration of nase into the Pielach started 10 days before spawning. Spawning took place from 27th to 29th of April. Six spawning grounds were monitored, four below the first continuum disruption at Spielberg and two smaller ones upstream (Table 1).

Four spawning habitats of barbel were observed 16 days later at 15th and 16th of May. Three of them were located downstream Spielberg. Barbels in this area migrated from the Danube into the River Pielach 14 days before spawning. Both barbel and nase move to different spawning sites, which have different distances to the main river (Table 1).

Table 1: Number of individuals and size of spawning habitats (sorted by distance to the Danube River).

Site	Date	Distance to Danube [m]	N individuals observed	Spawning habitat size [m²]
Nase habitat 1	28.04.1997	400	90	9 x 7 = 63
Nase habitat 2	29.04.1997	500	80	7.5 x 5.6 = 42
Barbel habitat 1	15.05.1997	650	150	30 x 8 = 240
Barbel habitat 2	15.05.1997	950	180	50 x 15 = 750
Nase habitat 3	27.04.1997	1200	240	28 x 3.5 = 98
Nase habitat 4	27.04.1997	1300	750	30 x 7 = 210
Barbel habitat 3	15.05.1997	1500	250	25 x 27 = 675
Migration barrier Spielberg		1900		
Barbel habitat4	16.05.1997	2350	30	20 x 8 = 80
Nase habitat 5	27.04.1997	3300	20	25 x 4 = 100
Nase habitat 6	27.04.1997	5000	70	14.5 x 6 = 87

The shortest migration distance to spawning grounds from the Danube into the Pielach was 400 meters. Theoretically, it was possible to move about 2 km upstream; at this point a 3 m high weir prevented further upstream migration. Below this barrier the largest spawning habitat was observed. Due to the migration from one of the last two free flowing sections of the Danube in Austria more than 1000 nase and more than 500 barbels entered the Pielach River and spawned in dense shoals.

Three smaller habitats were found within the next 3 km (Table 1). In this part only 30 spawning barbels and 90 spawning nases originating from the population resident in this river section could be identified.

Before and after spawning fishes occupied deeper habitats (pools) very close downstream of the spawning sites.

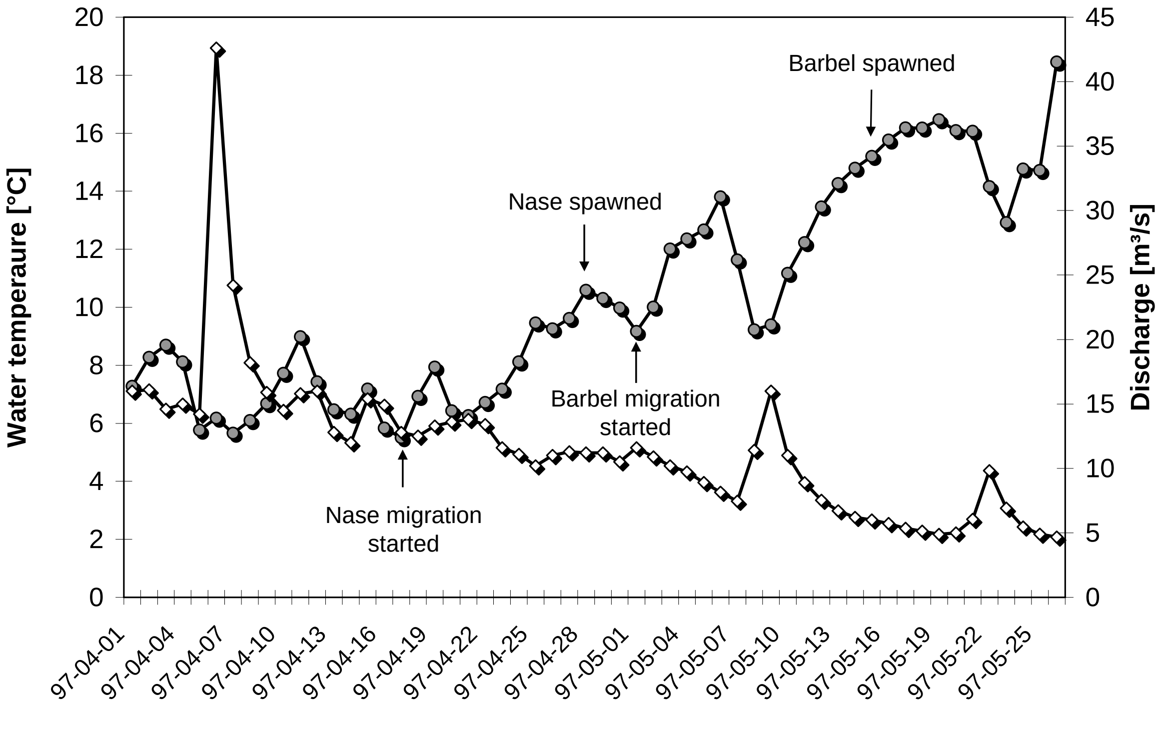

Figure 4: Daily mean water temperature (grey dots) and discharge (white symbols) and its relation to start of spawning migration and spawning activity.

The daily mean water temperature was 5.5 °C and the discharge was 12.5 m³/s in the Pielach River when the first nase entered from the Danube (Figure 4). They spawned 11 to 12 days later. At that time the water temperature increases up to 9.6 to 10.8°C and the discharge decreased by 2 m³/s.

When barbel started migration the water temperature increased from 10 to 14°C and discharge decreased from 10.5 to 7.5 m³/s. Two weeks later during spawning period the water temperature was 16°C with a discharge around 6 m³/s.

The variation in measured microhabitat variables was larger for the available habitat than for spawning habitat. The water depth was ranging from 4 to 290 cm, with a mean depth of 57 cm (Table 2). Mean velocity ranged from 0 to 163 cm/s, averaging 36 cm/s and the median diameter of substrate particles ranged from 2 to 6 cm, whereas up to 55 % loose substrate occurred. At 70 % of all microhabitats there was no cover or shading. The most dominating cover was overhanging vegetation. All microhabitat variables differ between available and spawning habitat (Mann Whitney *U*-Test, p=0.00 for all variables but broken water surface p=0.038).

Table 2: Mean (SD, median and range in parentheses) values of microhabitat variables in available habitat (Pielach River) and in nase and barbel spawning grounds. Presence of embeddedness, flow protection, shading, visual protection and cover structure are binary variables (1=present, 0=absent); the data are the proportions of sites that had a value of 1. Cover indicates the presence or absence of any cover structure type like overhanging or submerged vegetation, undercut bank, boulders or broken water surface. Asterisks indicate statistically significant between available habitat, nase and barbel spawning sites ($p < 0.05$).

Microhabitat variable	Available habitat (n=582)		Used habitat nase (n=1250)		Used habitat barbel (n=673)	
Water depth [cm]	57 ± 44*	(4-290)	34 ± 16	(16-83)	37 ± 16	(12-88)
v-bottom [cm/s]	22 ± 19*	(0-100)	67 ± 14	(20-97)	49 ± 15	(16-96)
v-mean [cm/s]	36 ± 27*	(0-163)	96 ± 13	(63-131)	65 ± 18	(20-107)
Substrate [cm]	2-6 ± 19.8*	(< 0.2-> 40)	2-6 ± 0	(2-20)	0.2-2 ± 0	(0.2-6)
Embeddedness	0.45 ± 0.50*	(0-1)	0 ± 0	(0-1)	0 ± 0	(0-0)
Flow-protection	0.07 ± 0.25	(0-1)	0 ± 0	(0-0)	0 ± 0	(0-0)
Shading	0.32 ± 0.47*	(0-1)	0.66 ± 0.47	(0-1)	1 ± 0	(1-1)
Visual-protection	0.10 ± 0.30	(0-1)	0 ± 0	(0-0)	0 ± 0	(0-0)
Cover	0.32 ± 0.47*	(0-1)	0.66 ± 0.47	(0-1)	0.75 ± 0.43	(0-1)
Overhanging vegetation	0.22 ± 0.42*	(0-1)	0.60 ± 0.49	(0-1)	0.75 ± 0.43	(0-1)
Submerged vegetation	0.06 ± 0.24	(0-1)	0 ± 0	(0-0)	0 ± 0	(0-0)
Undercut bank	0.01 ± 0.08	(0-1)	0 ± 0	(0-0)	0 ± 0	(0-0)
Rip rap	0.01 ± 0.02	(0-1)	0 ± 0	(0-0)	0 ± 0	(0-0)
Boulder	0.01 ± 0.07	(0-1)	0 ± 0	(0-0)	0 ± 0	(0-0)
Broken water surface	0.02 ± 0.14*	(0-1)	0.06 ± 0.24	(0-1)	0 ± 0	(0-0)

Correlation between all microhabitat variables and also for the available habitat and the spawning grounds was tested separately. The highest significant correlation was between mean and bottom velocity (nase r=0.64, barbel r=0.85, available habitat r=0.85) and between overhanging riparian vegetation and shading (nase r=0.88, available habitat r=0.77). The available habitat data showed no further correlations with r>0.4. There were some negative correlations with r<-0.4 for separate analyses of the spawning habitats especially for water depth and substrate (barbel r=-0.46), water depth and cover (nase r=-0.67, barbel r=-0.73), for water depth and shading (nase r=-0.67).

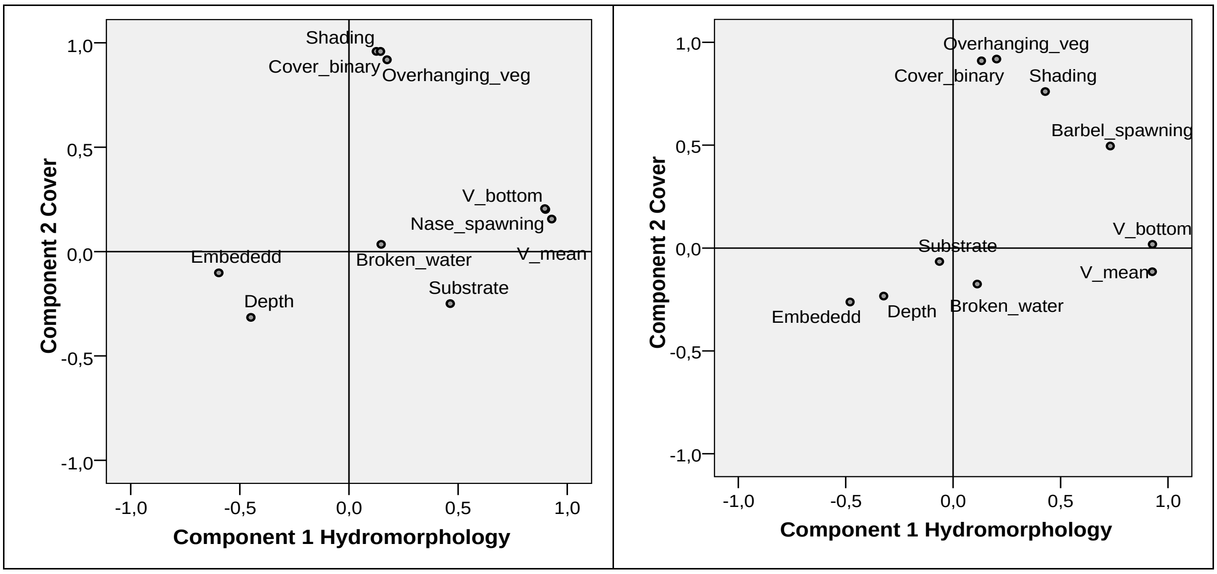

Figure 5: PCA component plot in rotated space (Varimax with Kaiser Normalisation) describing relation between microhabitat variables and spawning habitat for nase (left) and barbel (right). Presence of embeddedness (Embededd), shading (Shading), overhanging riparian vegetation (Overhanging_veg) cover (Cover_binary) and fish spawning habitat (Nase_spawning, Barbel_spawning) are binary variables (1=present, 0=absent);constant variables (i.e. undercut banks, are not considered in PCA).

For nase the PCA (Figure 5 left) indicated that principal component 1 (Hydromorphology) explained 42 % of the variance in spawning habitat utilization. Component 1 mainly consisted of mean and bottom water velocity (0.93 and 0.90), substrate and embeddedness (0.47 and -0.60) and water depth (-0.45). In addition 21 % of the variance was explained by principal component 2 (Cover), mainly through cover (overhanging vegetation) and shading (0.92 and 0.96).

For barbel the PCA (Figure 5 right) indicated that principal component 1 explained 39 % of the variance in spawning habitat utilization. Component 1 mainly consisted of mean and bottom water velocity (both 0.93), water depth (-0.33) and embeddedness (-0.48). In addition 17 % of the variance was explained by principal component 2, mainly through cover (overhanging vegetation) and shading (0.92 and 0.76).

For both the difference in variance explained by principal component 1 and 2 emphases the importance of flow velocities - to a lesser extend substrate or water depth in generating variation in spawning habitat utilization. Nevertheless the importance of vegetation and shading along a river especially for barbel was also clearly demonstrated.

Additional statistical tests (Mann Whitney *U*-Test) showed significant differences between nase and barbel spawning habitat use for all common occurring microhabitat variables (p=0.000) (Table 2). Effect size (η^2) explained the relative importance of mean velocity (0.70), substrate size (0.65), bottom velocity (0.54) and shading (0.39). Dominating available habitat (index > 0.5) can be characterised as: Mean velocity 21 - 50 cm/s (Figure 6), bottom velocity 1 - 30 cm/s; water depth 16 - 60 cm, substrate size 0.2 - 20 cm and about 50 % of the river is not embedded (Figure 7). Most of the river has no cover structure and only about 30% has shading (Table 2, Figure 7).

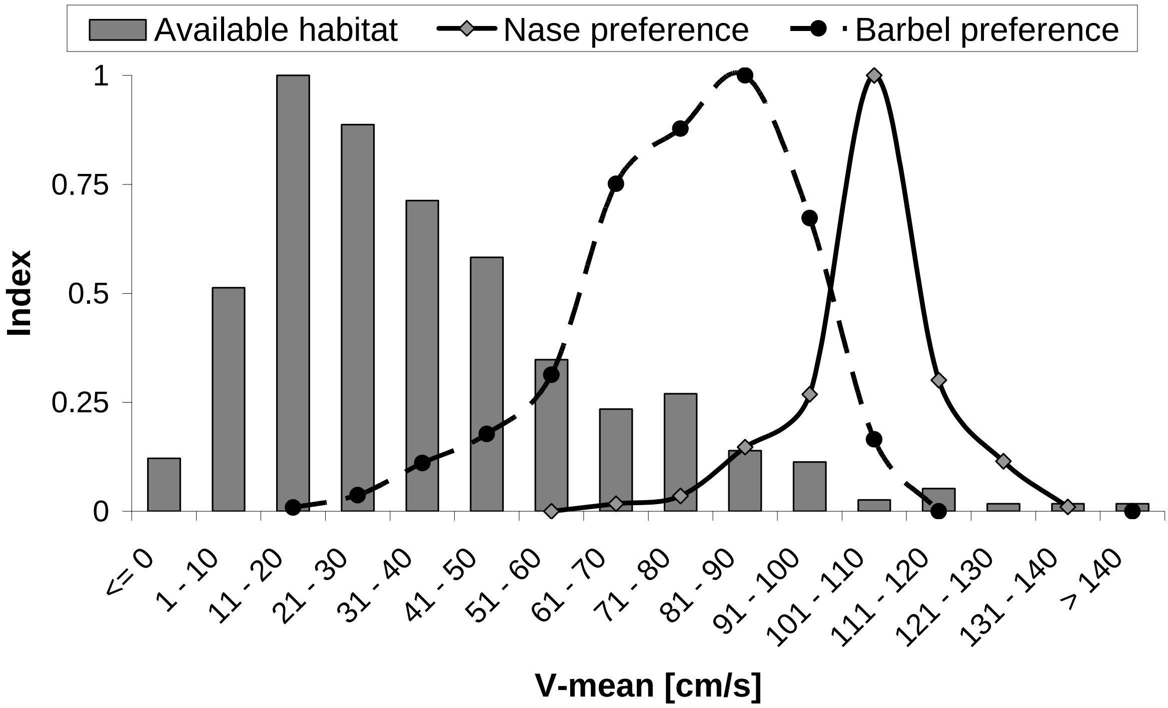

Figure 6: Spawning habitat preference curves of mean velocity for nase (n=1250) and barbel (n=673) and frequency-of-use index of mean velocity in available habitat (n=582).

The results showed on the one hand correlations between different microhabitat variables, but on the other hand significant differences between spawning habitat utilization of nase and barbel. Therefore, preferences describing nase and barbel spawning habitats are directly graphically compared (Figure 6 and 7). Habitat preference curves for mean water velocity indicated that very high mean water velocities, i.e. nase 100 -110 cm/s, barbel 61 – 100 cm/s, were preferred for spawning site construction. Nase uses riffles with very low water depth (16 - 30 cm) and barbel sites a little deeper. Due to the shallow water the bottom velocity is highly correlated with the mean velocity and preferences are also very high (51 – 60 cm/s for barbel and 81 – 90 cm/s for nase). Nase uses coarser substrate (2 – 6 cm) than barbel and both prefer loose substrate.

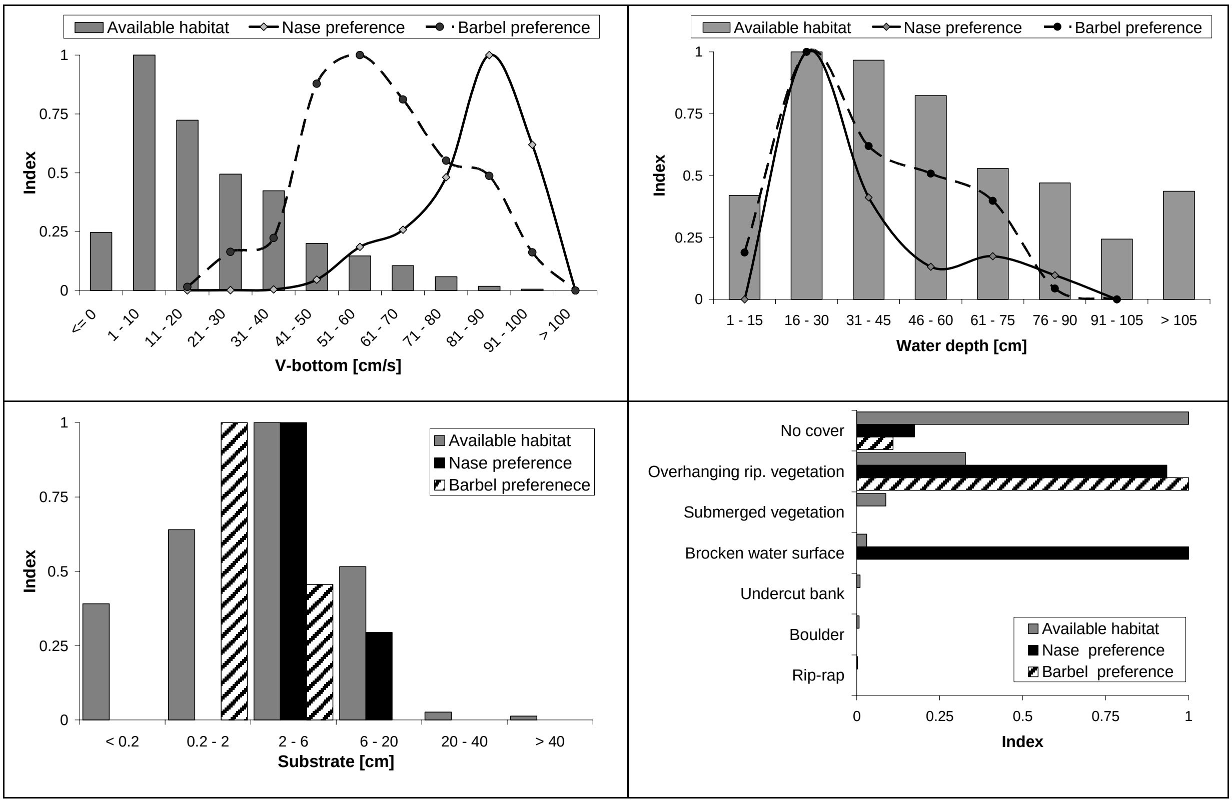

Figure 7: Spawning habitat preference curves of bottom velocity, water depth, substrate size and cover structure for nase (n=1250) and barbel (n=673) and frequency-of-use description of the available habitat (n=582).

Discussion

In this paper two rheophilic cyprinids nase and barbel were investigated that were once the dominant species in several mid-sized and larger European rivers (Schiemer and Waidbacher 1992). Their life-history patterns and habitat requirements make them good indicators of the ecological quality and the structural properties of river systems (Keckeis et al. 1996). The increasing hydromorphological pressures in many river systems were accompanied by a drastic decline in lithophilic cyprinids (e.g. Penaz 1996, Ovidio 2008). Nase and barbel are therefore indicator species for habitat quality in the lower rhithral and upper potamal zones of European river systems. Recent European wide studies showed that both indicator species are very rare nowadays (Melcher et al. 2007). Moreover, in the major part of their distribution area, both nase and barbel were also affected by persisting water pollution, reduction in food resources, changes in hydrological regimes and changes in riverbed morphology at spawning grounds (Penaz 1996).

Although information on spawning behaviour is very limited, some studies have addressed it for nase (Huber and Kirchhofer 1998, Ovidio and Philippart 2008) and barbel (Baras 1994, Baras et al. 1994 and 1996, Penaz 1996).

In our study the initiation of migration to spawning sites was linked to an increase of water temperature (10°C for nase and 16°C for barbel) and a decrease of discharge. Both species occupy fast flowing habitats (riffles) not very deep with coarse substrate on the one hand.

This study documented distinct variation in nase and barbel preferences of spawning habitats. Significant differences between nase and barbel spawning habitat were related to water velocity, substrate and cover structure, particularly shading. Although both species are classified as lithophilic spawners their spawning requirements are very different which has to be considered in defining restoration targets for lowland rivers inhabited by those species.

Velocity, water depth or temperature and spawning period were comparable to those reported in previously published studies. Baras and Philipart (1999) describe that barbel spawning was initiated when mean water daily minimum temperature reached 14.0°C and at maximum 18.0°C. For nase Rakowitz et al (2008) found that at the spawning area the highest abundance occurred at 9.5°C. Penaz (1996) also found out, in accordance to us, that nase spawning sites are located on shallow (10-30 cm) riffles with a rather rapid current (0.6 – 1.5 m/s). Information on riparian vegetation structures are missing.

Our results show the essential role of cover structure in physical habitat studies. Especially vegetation along a river shoreline and resulting shadow is a very important for spawning sites of barbel and nase. In channelised rivers the flow often is increased and the water is shallower compared to natural rivers which might create potential spawning habitats for nase and barbel. However, the lack of riparian vegetation and shading as well as strong embeddedness of the substrate disqualifies channelised rivers as suitable spawning habitats. Furthermore, defining minimum flow requirements in barbel zone should include besides traditional parameters (Hauer et al. 2007 and 2008), i.e. flow velocity, water depth and substrate, additional essential parameters, i.e. shading (riparian vegetation) and embeddedness, as demonstrated in our study. This places the minimum flow issues as a component of overall hydromorphological river restoration and broadens the applicability of physical habitat models (e.g. Oliveira et al 2004). As most European rivers are affected by both morphological and hydrological pressures (Schinegger et al. 2009) river restoration strategies have to strive for an integrated approach covering both hydromorphological and additional essential habitat attributes. Further studies and more sophisticated analytic tools are necessary for quantification of biological consequences of multi-impacted environments (Le Pichon et al. 2006a, Souchon et al. 2008).

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References

Ahmadi-Nedushan B., A. St-Hilaire, M. Berube, E. Robichaud, N. Thiemonge and B. Bobee 2006: A Review of statistical methods for the evaluation of aquatic habitat suitability for instream flow assessment. River Res. Applic. 22: 503–523.

Balon, E.K., 1975b: Reproductive guilds of fishes: a proposal and definition. L. Fish. Res. Board Can. 32: 821-864.

Baras, E. 1992: A study of time and space utilisation strategies in the common barbel Barbus barbus (L.). Cahiers d'Ethologie 12:125–442.

Baras, E. 1994: Constraints imposed by high densities on behavioural spawning strategies in the barbel, Barbus barbus. Folia Zoologica 43, 255–266.

Baras, E. 1997: Environmental determinants of residence area selection by Barbus barbus in the river Ourthe. Aquatic Living Resources 10:195–206.

Baras, E., H. Lambert, and J. C. Philippart. 1994: A comprehensive assessment of the failure of Barbus barbus spawning migrations through a fish pass in the canalized river Meuse (Belgium). Aquatic Living Resources 7:181–189.

Baras, E., Philippart, J.-C. & Nindaba, J. 1996: Importance of gravel bars as spawning grounds and nurseries for European running water cyprinids. In Proceedings of Ecohydraulics 2000, Vol. A (Leclerc, M. et al., eds), pp. 367–368. Que´bec City: INRS-Eau.

Bretschneider, H., Lecher, K., and Schmidt, M. (Eds.). 1993. Taschenbuch der Wasserwirtschaft, seventh edn. Verlag Paul Peary, Hamburg.

Bühl, A. and P. Zöfel, 2004: SPSS Version 12, Einführung in die moderne Datenalyse unter Windows. Addison Wesley Verlag, München: 734.

Hauer C, Unfer G, Schmutz S, Habersack H 2007: The importance of morphodynamic processes used as spawning grounds during the incubation time of nase (Chondrostoma nasus). Hydrobiologia 579:15–27

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

Huber, M.; A. Kirchhofer, 1998: Radio telemetry as a tool to study habitat use of nase (Chondrostoma nasus L.) in medium sized rivers. Hydrobiologia 372, 309–319.

Huber, M.; A. Kirchhofer, 2001: Reproductive sucess of nase (Chondrostoma nasus L.) and its influence on population dynamics. Large Rivers Vol 12, No 2-4, 307 - 330

Huet M 1949: Apercu des relations entre la pente et les populations piscicoles des eaux courantes. Schweiz Z Hydrol 11:332–351

Ivlev V.S., 1961: Experimental Ecology of the Feeding of Fishes. New Haven, Yale University Press.

Keckeis, H.; Frankiewicz, P.; Schiemer, F., 1996: The importance of inshore areas for spawning nase Chondrostoma nasus (Cyprinidae) in a free-flowing section of a large river (Danube, Austria). In: The ecology of large rivers. Archiv. Hydrobiol. D. W. Sutcliffe (Ed.). 10 (Suppl. 113), 51–64.

Kirk, R. E. 1982: *Experimental design: Procedures for the behavioral sciences* (2nd ed.). Belmont, CA: Brooks/Cole.

Le Pichon C, Gorges G, Boe¨t P, Baudry J, Goreaud F, Faure T. 2006a: A spatially explicit resource-based approach for managing stream fishes in riverscapes. Environmental Management 37: 322–335.

Lelek, A.; Penaz, M., 1963: Spawning of Chondrostoma nasus (L.) in the Brumokva River. Folia Zool. 12, 121–134.

Melcher, A, S. Schmutz, G. Haidvogl, K. Moder 2007: Spatially based methods to assess the ecological status of European fish assemblage types. FISHERIES MANAG ECOL, 14, 453-463.

Moir, H. J. and. Pasternack G. B 2009: Substrate requirements of spawning Chinook salmon (Oncorhynchus tshawytscha) are dependent on local channel hydraulics. River Research and Applications. Pages n/a. © 2009 John Wiley & Sons, Ltd.

Northcote, T.G. 1984: Mechanisms of fish migration in rivers. In: McCleave, J.D., Arnold, G.P., Dodson, J.J. & Neill, W.H., eds. Mechanisms of migration in fishes. New York: Plenum Press, 317–355.

Oliveira J. M., M. T. Ferreira, A. N. Pinheiro and J. H. Bochechas 2004: A simple method for assessing minimum flows in regulated rivers: the case of sea lamprey reproduction. Aquatic Conserv: Mar. Freshw. Ecosyst. 14: 481–489

Ovidio M. and J. C. Philippart 2008: Movement patterns and spawning activity of individual nase Chondrostoma nasus (L.) in flow-regulated and weir-fragmented rivers. J. Appl. Ichthyol. 24 (2008), 256–262

Parasiewics P. and M.J. Dunbar 2001: Physical habitat modelling for fish – a developing approach. Large Rivers Vol. 12, No 2-4, 239-268

Penaz, M. 1996: Chondrostoma nasus-its reproduction strategy and possible reasons for a widely observed population decline-a review. In: Conservation of endangered freshwater fish in Europe. A. Kirchhofer and D. Hefti (Eds). Birkha¨ user Verlag, Basel, Switzerland, pp. 278–285.

Rakowitz G, Berger B, Kubecka J, Keckeis H. 2008: Functional role of environmental stimuli for the spawning migration in Danube nase Chondrostoma nasus (L.). Ecology of Freshwater Fish 2008: 17: 502–514.

Raleigh, R. F., L. D. Zuckerman, and P. C. Nelson 1986: Habitat suitability index models and instream flow suitability curves: brown trout. U.S. Department of the Interior, Fish and Wildlife Service, National Ecology Center, Biological report 82 (10.124)

Santos, J. M. and M. T. Ferreira (2008) Microhabitat use by endangered Iberian cyprinids nase Iberochondrostoma almacai and chub Squalius aradensis. Aquat. Sci. 70 (2008) 272 – 281.

Schiemer, F. & Waidbacher, H. 1992: Strategies for conservation of a Danubian fish fauna. In: Boon, P.J., Calow, P. & Petts, G.E., eds. River conservation and management. Chichester: John Wiley & Sons Ltd, pp. 363–382.

Schinegger R., Melcher A., Trautwein C. & S. Schmutz (2009): Detecting patterns and relationships of human pressures in European rivers. International Association of Hydraulic Engineering and Research, 33rd IAHR Congress: Water Engineering for a Sustainable Environment; 4702 – 4709. ISBN: 978-94-90365-0

Shirvell CS. 1989. Ability of PHABSIM to predict chinook salmon spawning habitat. Regulated Rivers Research and Management 3: 277–289.

Sokal, R. R. & F. J. Rohlf 2001: Biometry: The Principles and Practice of Statistics in Biological Research. W. H. Freeman and Company, New York.

Souchon Y, C. Sabaton, R. Deibel, D. Reiser, J. Kershner, M. Gard, Ch. Katapodis, P. Leonard, N. Leroy Poff, W. Miller, B.L. Lamb 2008: Detecting biological responses to flow management: missed opportunities, future directions. River. Res. Applic. 24: 506–518

Stanford J A and J V Ward 1988: The hyporheic habitat of river ecosystems. Nature Vol. 335. 64 - 66

Tabachnick, B. G., & Fidell, L. S. 1989: *Using multivariate statistics* (2nd ed.). New York: Harper & Row.

Wollebaek, J, R Thue, and J Heggenes 2008: Redd site microhabitat utilization and quantitative models for wild large brown trout in three contrasting boreal rivers. North American Journal of Fisheries Management 2008; 28: 1249-1258

Zbinden, S.; Maier, K. J., 1996: Contribution to the knowledge of the distribution and spawning grounds of Chondrostoma nasus (Pisces, Cyprinidae) in Switzerland. In: Conservation of endangered freshwater fish in Europe. A. Kirchhofer and D. Hefti (Eds). Birkhäuser, Verlag, Basel, Switzerland, pp. 287–297.

Zitek, A. & S. Schmutz 2004: Efficiency of restoration measures in a fragmented Danube/tributary network. Proceedings of the fifth international conference on ecohydraulics - aquatic habitats: analysis and restoration (12–17.09.04), Madrid. IAHR: 652–657.

Zitek, A. S. Schmutz & M. Jungwirth 2008: Assessing the efficiency of connectivity measures with regard to the EU-Water Framework Directive in a Danube-tributary system. Hydrobiologia 609:139–161

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Processes of collating a European fisheries database to meet the objectives of the European Union Water Framework Directive

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Abstract FIsh Database of European Streams, a common database for the FAME project, was merged using existing data on electric fishing and environmental data. FIsh Database of European Stream is a relational database with eight tables. Metrics based on classification of fish species into guilds were calculated, and provided in separate tables. FIsh Database of European Stream contained information about 150 freshwater fish species, from 12 countries, 17 ecoregions, 40 main river regions, 2651 rivers and 8228 sites. Examples of data coverage and use are given. Relationships between environmental variables were illustrated using principal component analysis, which resulted in three environmental components – latitude, size and altitude. Environmental component scores were correlated with fish metrics used in the European Fish Index. Results exemplify how fish guilds reflect gradients in environmental variation. Benefits and problems concerning standardisation and data availability at the global level are discussed.

KEYWORDS: environmental variables, fish abundance, fish communities.

Introduction

The European Union Water Framework Directive (WFD; 2000/60/EEC) requires Member States to assess the ecological status of rivers, lakes and transitional waters using ecological criteria, including fish. To meet these needs, a standard methodology that provides an unbiased means of quantifying ecological status and a measure of response to human pressures is required across the Member States. Fish community data are known to be a good indicator of ecological health (Karr & Chu 1999) and are routinely monitored by many jurisdictions. Unfortunately, in Europe these data are collected by a multitude of agencies and institutions using diverse array of gears and methodologies (Cowx & Lamarque 1990), often with differing objectives and accuracy. The methodologies vary between river type and

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topography, as well as available resources. Consequently, to develop a pan-European methodology for assessment of the ecological status of surface water bodies using fish as the biological criteria, requires a strategic approach to data collection, collation and management.

This paper describes the rational for, contents of and standardisation procedures to develop a pan-European database (named FIDES – FIsh Database of European Streams) for fisheries that supported the FAME project (http://fame.boku.ac.at). There are few other cases where common databases have been put together to serve as a basis for developing a method for assessment of ecological status. One European example, containing biological as well as environmental data, is the AQEM database with focus on stream benthic invertebrates (AQEM 2002; Hering, Moog, Sandin & Verdonschot 2004). Consequently, this represents an opportunity to highlight the pitfalls and procedures for establishing such a comprehensive database. Examples of the coverage of the biological data over large geographical scales are given, as well as how these data may be used.

Data requirements

To develop a European Fish Index for running waters requires a common source of data for analyses and evaluation. The data needed include standardised measures of fish density, environmental information, quantification of different human impacts with respect to fish and information on historical occurrences of fish species. After consultation with participating agencies and institutions, fisheries data collected during routine monitoring surveys supplemented by research programmes are the best available. In particular, data collected during electric fishing surveys seemed the most consistent because they were collected under a common platform, which has since become a European standard (CEN 2003). Electric fishing is a commonly used sampling method because fish are usually released alive after species have been identified and biometric measurements taken. Such non-destructive sampling methods are rare in the field of aquatic biology. Furthermore, the sample acquired can be semi-quantitative or quantitative depending on effort (Cowx 1995). Thus, not only species and length (age) composition, but also population density, may be quantified, and at comparatively low cost. The drawbacks are mainly a limitation to approximately 2 m depth as well as potential safety problems, because of the hazardous combination of water and electricity (Cowx & Lamarque 1990).

Although standard operating procedures exist for electric fishing (CEN 2003), the method can vary depending on objective, type of water and required precision in population abundance. For example, different sampling strategies, typically single run relative abundance vs successive removal absolute methods, are used to determine fish abundance and fishing is carried out wading upstream or from a boat or a combination of both (see Cowx 1995 for summary). Therefore, it was essential that the FIDES database contained methodological information such as date, geographical location, details of the fishing method, sampling strategy and sampled site characteristics. The problem of fishing intensity was resolved by only using data from the first run in successive removal surveys so it was comparable with single run data, as there exists a strong relationship between single run and multiple run estimates of fish abundance (Jones & Stockwell 1995). This approach was used to compare between stream types and methodologies. If less than 10 individuals were caught in the first run, the sampling was not included in the modelling data set (Pont, Hugueny and Rogers 2007). The area of sampling also needed to be standardised. The CEN standard states that when the aim of the sampling is explicitly to sample the whole fish community, electric fishing must be conducted over stream lengths of at least 10–20 times the stream width (Angermeier & Karr 1986; CEN 2003). The threshold for inclusion of data was a minimum of 100 m^2 sampled area (Pont, Hugueny & Rogers 2007).

Geography, climate and other environmental and biotic factors determine the occurrence and abundance of fish species. Physical and chemical conditions, e.g. conductivity (nutrients), depth, substrate and flow, also affect fish populations and probability of capture (Bohlin, Hamrin, Heggberget, Rasmussen and Saltveit 1989; Cowx 1990). Therefore, a comprehensive set of data with 209 parameters, such as geographical location, altitude, catchment area and other permanent features, was required (Fig. 1). The final choice of environmental variables was based on common practice, consistency in availability and studies combining biological and environmental data (e.g. Karr 1981; Oberdorff, Pont, Hugueny & Chessel 2001; Oberdorff, Pont, Hugueny & Porcher 2002; CEN 2003), as well as on the AQEM protocol (AQEM 2002).

Database structure, input procedures and quality assurance

A relational database (Hernandez 2003) was set up as a collection of eight tables with relations between key

Table Site	
Site_code	
Latitude	
Longitude	
Country_abbreviation	Geological_typology
Eco_region_no	Geological_formation
Subecoregion	Mean_air_temperature
River_type	Mean_Jan_temperature
Main_river_region	Mean_Jul_temperature
River_name	Gradient_slope
Site_name	Huet_zonation
National_map_code1	Other_zonation
National_map_code2	Stream_order
Size_of_catchment_class	Lakes_upstream
Size_of_catchment	Distance_to_lake
Width_flooded_area	Distance_from_source
Mean_discharge_class	Distance_to_mouth_class
Flow_regime	Distance_to_mouth
Altitude	Water_source_type

Table Fishing_occasion	
Site_code	
Latitude	
Longitude	
Date	
Reporter_code	
Numdate	(1) Land_use_river
Sampling_strategy	Urbanisation_river
Method	Connectivity_river
Day_Night	Land_use_segment
Number_of_runs	Urbanisation_segment
Runs_separated	Riparian_zone_segment
Water_temperature	Connectivity_segment
Conductivity_class	Connectivity_multiscale
Conductivity	Floodplain_lateral_movements_segment
Locality_length	Sediment_load_segment
Locality_width	Hydrological_regime_site
Fished_area	Natural_flow_pattern_site
Wetted_width	Natural_flow_quantity_site
Average_depth	Upstream_dam_site
Maximum_depth_class	Morphological_condition_site
Dominating_substrate	Salinity_site
Stop_nets_used	Toxic_acidification_site
No_of_anodes	Nutrients_organic_input_site
Type_of_anode	Introduction_of_fish_site
Size_of_anode	Impact_of_stocking_site
Type_of_current	Exploitation_site
Voltage_used	Fauna_impact_site
Wattage_used	Flora_impact_site
Mesh_size	Weed_cutting_site
(2) Reference_site	Status_ecoregional
Calibration_dataset	Status_European_spatial
Modelling_dataset	Status_modelling
Fish_type_ecoregional	Local_IBI_method
	Local_IBI_score
	Local_status_classification

Table Catch
Site_code
Latitude
Longitude
Date
Species
Run1_number_all
Run1_number_0_plus
Run2_number_all
Run2_number_0_plus
Run3_number_all
Run3_number_0_plus
Run4_number_all
Run4_number_0_plus
Total_number_all
Total_number_0_plus
Total_biomass
Biomass_estimate
Total_abundance
Abundance_estimate
Estimated_efficiency

Table Length
Site_code
Latitude
Longitude
Date
Species
Type_of_data
Length_type
Length

Table Length_class
Site_code
Latitude
Longitude
Date
Species
Type_of_data
Length_type
Length_class_min
Length_class_max
Number

Historical_data
Site_code
Period_start
Period_end
Period_string
Species
Status_scale_class
Abundance_class

Table Reporter
Reporter_code
Responsible_person
Company_University
Institute_Agency
Street_POB
City
State_Province
Postal_code
Country
Email

Table Taxa_and_guilds	
Species	
Fides_occure	(Guild classification 16 variables)
Tolerance	
Habitat_feeding	(Sentinel species 14 variables)
Habitat_rheo	
Reproduction	(Native or alien per Main River Regions 40 variables)
Feeding	
Migration	
Longevity	

Figure 1. Variables included in FIsh Database of European Streams. Variables at the top of each table written with grey background (names centred) were key variables in the relation database. Variables written in boldface are mandatory variables, the others optional. Variables written in italics were categorical variables with pre-defined categories. In the Fishing occasion table, the variables within box (1) are human impact variables, and within box (2) derived variables, or related information preceding analyses.

variables (Fig. 1). In total, 209 variables were included. Quality assurance was essential to fulfil quality requirements and was achieved by providing an empty database with all input tables. Furthermore, metadata were provided to describe variables, together with a manual to aid data input. This input database was made using Microsoft® Access 2000 and comprised several automated quality control tools. These included, for example, checking for missing mandatory variables, checking correct longitude and latitude, total number of each species caught matches total catch and inaccuracies in length of individual fish. The main principles for data quality controls were to prohibit double entries, to ensure traceability in connection to site, time and species, as well as correct any easily recognised errors. Quality control measures were updated frequently, as unforeseen errors occurred during data delivery.

National data contacts were nominated to deliver data that fulfilled the project requirements, and were representative of the variation within each country. The national data contacts were responsible for the quality control of the national data, as a complete systematic screening of FIDES was not possible because of time constraints and these persons should better understand the idiosyncrasies of the data. Data were collected from one or several sources in the country. For example, Germany collected national data from 11 different federal states, while Sweden collected data from one national register (Swedish Electrofishing RegiSter – SERS). Data were again quality controlled using procedures specific to each country, as well as the control tools, before the national data set was merged into FIDES. The database manager controlled 51 versions of national data (mean $n = 4$ per country), until all national data sets met sufficient quality requirements to be merged into FIDES, set up as a relational database implemented on a Microsoft® SQL Server 2000 database management system.

To facilitate data management and sharing, a series of input and export queries were developed using Data Transformation Services (DTS). These utilities allow import and export between different data sources and the capability to provide refined data back to the donor countries.

Summary statistics available in FIDES

Electric fishing sampling methodology

Fishing was carried out by wading on 71% of occasions and by boat on 29%. Boat fishing was not performed in Greece, Spain and Sweden. Generally deeper, wider waters with low gradient were fished using a boat. All fish sampling was performed during the day, except on nine occasions (0.1%). The maximum likelihood method (Zippin 1958) is commonly used to quantify fish abundance. If a single pass (run) is carried out, the sample may be regarded as semi-quantitative but this approach allows comparisons among streams and methodologies. Only one fishing pass was undertaken on 52% of occasions, two passes on 24% and three or more passes on 24% of fishing occasions. Boat fishing typically applied single passes on 62% of occasions while wading used single (48%) and multiple runs almost equally. Alternating current (AC) was used on 7% of occasions, direct current (DC) on 75% and pulsed DC on 18%. The use of AC was restricted to boat fishing and DC was the most commonly used when wading (84% of occasions). The most common anodes used when wading, were ring (86%) or rectangular (13%) design. These designs were the main alternatives used during boat electric fishing (ring – 62% and rectangular – 28% of occasions); boom or other types of anodes were only used on 10% of boat fishing occasions. Stop or block nets upstream or downstream of the fished location were only used in 17% of occasions. Stop nets were used up to an average wetted width of 100 m, but generally nets were used in small rivers (median wetted width 7.5 m). The mesh size of the landing net ranged from 1 to 15 mm, and was 4–6 mm in 58% of all fishing occasions.

Although it is recommended (CEN 2003) that the stream length sampled should be at least 20 times the wetted width, this was only the case for 40% of sampling occasions. The reason for the recommendation is to ensure that the sample should be representative with regard to the number of species, species composition and capture of sufficient individuals. It was generally the wider and deeper sites (mean width 7.6 m and depth 1.0 m) where shorter lengths of river were sampled.

Metrics data

According to the WFD, three major categories of data need to be examined to determine ecological status: species composition (including metrics related to trophic composition, reproduction and lifestyle), fish abundance and age-length structure. These were included in FAME assessment methodologies (see Pont *et al.* 2007; Melcher, Schmutz, Haidvogl and Moder 2007) using specific metrics based on fish population/community structure and abundance. The metrics selected are described by Noble, Cowx, Goffaux and Kestemont (2007) and Roset, Grenouillet, Goffaux, Pont and Kestemont (2007), of which several are derived in the FIDES database (Table 1).

Only data from first run electric fishing were used to develop the European Fish Index (Pont *et al.* 2007) to allow comparisons between single-run sampling with multiple-run sampling. However, in some cases only total estimated biomass (kg ha^{-1}) was provided, so the biomass for the first run was determined based on the assumption that the ratio between total abundance (total_number_all) and abundance first run (run1_number_all) was the same as for total biomass (total_biomass) and biomass in the first run (first run biomass).

Differentiation between alien, native and all species was made per main river region in different versions of the Taxa and Guilds table (Fig. 1, Noble, Cowx, Goffaux & Kestemont 2007). Guild classification was checked, harmonised with the species table in FIDES and reformatted to fit the format of the Taxa and Guilds table of FIDES. Each guild category (tolerance, habitat, migration, etc.) represents a variable and species are now classified within each category (tolerant, intolerant, etc.; Table 1).

Metrics comparing historical and present occurrence of native fish species were added and could be used for sites/fishing occasions where full sets of historical data (= information about total species composition) were provided. (The variable 'number of historical species' was added to the metric table only for calculating historical metrics. This was made for 22% of the sites, most of them in Austria, France and Belgium.). For historical metrics, the percentage of historical species in each guild was computed. One additional historical metric, comparison of historical and actual occurrence of diadromous species as defined by presence within a river basin, was added. The classification table of 45 sentinel species, i.e. key species, was also completed and integrated into the Taxa and Guilds table (Fig. 1).

A software routine was developed and included the following steps: (1) general check of data consistency (main river regions, guilds, fish species names and sentinel species, species occurrence in catch); (2) calculation of the biomass of the first run; (3) selection of native and alien species at each site; (4) calculation of the metrics table for all sites; (5) calculation of the

Table 1. Overview of computed metrics included as separate tables in FIsh Database of European Streams

Guild		Number of all species	Number of native species	Percent native species	Density (n ha^{-1})	Biomass (kg ha^{-1})	Biomass 1. Run (kg ha^{-1})	Species number	Percent species of number of native	Individuals (n ha^{-1})	Percent individuals of density	Biomass (kg ha^{-1})	Percent biomass of biomass	Biomass 1. Run (kg ha^{-1})	Number of historical species	Percent historical of native species	Presence YOY	Density YOY (n ha^{-1})	Percent of YOY
Overall composition	All species	x																	
	Native species		x	x															
Abundance	All species				x	x	x												
	Native species				x	x	x												
	Alien species				x	x	x												
Tolerance	Intolerant							x	x	x	x	x	x	x		x			
	Tolerant							x	x	x	x	x	x	x		x			
Habitat	Water column							x	x	x	x	x	x	x		x			
	Benthic							x	x	x	x	x	x	x		x			
	Rheophilic							x	x	x	x	x	x	x		x			
	Limnophilic							x	x	x	x	x	x	x		x			
	Eurytopic							x	x	x	x	x	x	x		x			
Reproduction	Lithophilic							x	x	x	x	x	x	x		x			
	Phytophilic							x	x	x	x	x	x	x		x			
Longevity	Long lived							x	x	x	x	x	x	x		x			
	Short lived							x	x	x	x	x	x	x		x			
Feeding	Piscivorous							x	x	x	x	x	x	x		x			
	Insectivorous/ invertivorous							x	x	x	x	x	x	x		x			
	Omnivorous							x	x	x	x	x	x	x		x			
Migration	Long distance							x	x	x	x	x	x	x		x			
	Potamodrom							x	x	x	x	x	x	x		x			
Historical 1	Status scale > 0														x	x			
Historical 2	Status scale = 7														x	x			
45 Sentinel species	Abundance				x		x												
46 Sentinel species	Age-length structure																x	x	x

Three versions were calculated: one for native, one for alien and one for all species (Noble *et al*. 2007).

metrics table for native species only; and (6) calculation of the metrics table for sentinel species. These summary statistics were made available as three tables for search and download within the web application; table Metrics_all, Metrics_native and Metrics_sentinel.

In general, mandatory variables were completed, but often the information had to be simplified into categories, or was based on operator expertise. Optional variables were more precise, but often only available for certain countries and/or a limited proportion of sites. Control and use of the data further showed that many environmental variables still contained at least a few errors, and some variables contained relatively high percentages of errors for certain countries. For example, nearly 5% of sites had errors in geographical coordinates because of data entry errors or confusion between different coordinate systems; these errors were concentrated in three or four countries.

Representativeness of FIDES in Europe

FIsh Database of European Streams contains information from 12 countries, 17 ecoregions, 40 main river regions, 2651 rivers and 8228 sites (Tables 2 and 3). The locations of sites were concentrated in the central western part of Europe, partly because of the distribution of FAME partners but also intensity of sampling programmes within countries. The 12 countries covered 72% of the area (2.9 million km^2) and 77% of the population of the EU-25 Member States. Nine of 18 ICES sea areas (ICES 2004) and eight out of Europe's 16 largest rivers were represented within FIDES (Table 4). Large rivers that drain into the Black Sea, Sea of Azov, White Sea, Barents Sea or Caspian Sea are not present in the FIDES database. FIsh Database of European Streams also does not include data for streams or rivers that drain into the Greenland Sea, Norwegian Sea, Inland Sea of the West Coast of Scotland, Bay of Biscay and Sea of Marmara.

A total of 150 species, out of 301 potential species occurring in European rivers (Noble *et al.* 2007), were recorded. The highest species richness was in the River Danube with 67 species, while only 12 species were recorded for the Douro on the Iberian Peninsula (Table 3). The mean number of caught taxa for all sites was 5.3 (SD: 3.7). The mean number at each site was correlated with the total number of species in the main river region (Spearman's $\rho = 0.341$, $P = 0.016$, $n = 40$). The most frequently occurring fish species was brown trout, *Salmo trutta* L., occurring at 47% of the sites. Although considered endangered at the European scale (Dekker 2003), European eel was frequently encountered at the investigated sites (33%). Mean abundance (individuals per hectare) for the sentinel species ranged between 73 for pike, *Esox lucius* L., and 2143 for Eurasian minnow, *Phoxinus phoxinus* (L.).

Table 2. Data included in FIsh Database of European Stream, for partner countries

	Table site		Table fishing occasion			Table catch	Table length/length class
Country	Number of cases	Mean percent valid cases for variables used in development of European Fish Index	Number of cases	Mean percent valid cases for variables used in development of European Fish Index	Percent sites with multiple fishing occasions	Number of recorded catches (species at fishing occasions)	Percent fishing occasions with provided length data
Austria	1053	100	1395	100	15	7595	99
Belgium Flanders	1042	100	1684	100	41	7769	39
Belgium Wallonia	120	100	158	100	12	1485	100
Germany	1854	62	2300	42	12	12 980	68
Spain	294	100	356	100	21	747	57
France	815	100	1584	100	48	12 420	0
Greece	56	100	83	100	23	200	73
Lithuania	253	100	355	100	23	2863	25
The Netherlands	662	100	976	88	26	5619	0
Poland	154	100	154	100	0	967	73
Portugal	232	100	232	95	0	997	0
Sweden	623	97	3765	100	80	10 888	93
United Kingdom	1070	100	2141	99	27	14 030	58
Total	8228	91	15 183	90	15	78 560	59

Table 3. Number of rivers, sites, mean, SD and total number of fish species in FIsh Database of European Streams by main river region, and representation of sites per main river region within Illies' ecoregions

Main river region	Number of rivers	Number of sites	Mean number of species	SD	Total number of species	Illie's ecoregion representation for sites occurring in main river region
Douro	29	51	3.8	1.5	12	Iberian Peninsula
Guadiana	13	19	6.0	2.0	17	Iberian Peninsula
North-East Atlantic Ocean	42	77	3.2	2.0	20	Iberian Peninsula
Tagus	69	85	5.1	2.6	22	Iberian Peninsula
Ebro	14	88	1.8	1.6	15	Iberian Peninsula; Pyrenees
Mediterranean Sea, Western Basin	92	206	2.2	1.6	22	Iberian Peninsula; Pyrenees
Aegean Sea	7	11	2.7	1.8	15	Helenic Western Balkan
Ionian Sea	29	45	2.0	1.7	22	Helenic Western Balkan
Garonne	72	97	6.0	4.3	41	Pyrenees; Western Plains; Western Highlands
North Atlantic Ocean	66	81	7.9	3.4	40	Pyrenees; Western Plains; Western Highlands
Mediterranean Sea, Western Basin, North Pyrenees	73	110	4.7	3.3	22	Pyrenees; Western Plains; Western Highlands; Alps; Italy
Rhone	67	93	6.2	6.1	42	Western Plains; Western Highlands; Alps
Loire	102	112	9.2	4.7	42	Western Plains; Western Highlands
Meuse	159	553	6.7	4.3	50	Western Plains; Western Highlands
Seine	62	92	9.5	4.0	38	Western Plains; Western Highlands
English Channel (Ecoregion Western Plains)	62	82	6.1	3.1	33	Western Plains
North Sea	309	1090	3.8	3.8	48	Western Plains; Central Plains
Rhine	294	1663	5.2	3.3	52	Western Highlands; Alps; Central Highlands; Central Plains
Danube	338	1141	4.9	3.8	67	Alps; Central Highlands; Hungarian Lowlands; Dinaric Western Balkan
Elbe	35	160	7.3	3.9	40	Central Highlands; Central Plains
Odra	19	104	7.5	4.1	37	Central Highlands; Central Plains
Wisla	6	32	7.8	5.7	34	The Carpathians; Eastern Plains
Weser	19	215	5.0	2.8	38	Central Plains
Skagerrak	45	80	2.8	1.7	29	Central Plains; Fenno-Scandian Shield
Kattegat Sound	46	97	2.8	1.4	30	Central Plains
Baltic Sea	236	550	3.3	2.6	49	Central Plains; Baltic Province; Fenno-Scandian Shield; Borealic Uplands
Gulf of Riga	4	13	5.8	3.2	23	Baltic Province
Nemunas	85	211	8.1	3.3	37	Baltic Province
Anglian Coast	39	304	6.6	2.3	40	Great Britain
Bristol Channel	8	16	6.1	2.8	16	Great Britain
English Channel (Ecoregion Great Britain)	30	133	4.8	2.9	26	Great Britain
Great Ouse	3	19	8.7	2.2	20	Great Britain
Irish Sea	103	178	3.5	1.5	15	Great Britain
Medway	4	11	5.5	2.3	17	Great Britain
Mersey	9	48	4.2	2.0	17	Great Britain
Severn	33	118	5.8	2.8	26	Great Britain
Tees	1	26	6.7	3.1	16	Great Britain
Thames	24	97	8.0	2.8	24	Great Britain
Trent	20	75	6.2	3.2	26	Great Britain
Yorkshire Ouse	7	45	6.9	3.5	24	Great Britain
Total number (40 main river regions)	2651	8228	5.3	3.7	150	17

Main river regions were defined as river catchments larger than 25 000 km^2, or as marine regions where several rivers confluence into the sea. Mean and SD represent 8228 fishing occasions at sites. If the fishing occasion at the site was used as calibration data for the European Fish Index (Pont *et al.* 2007) that fishing occasion was used, otherwise one fishing occasion was randomly selected for each site.

Table 4. Principal component analysis results (rotated loadings, based on Spearman rank correlation matrix) of environmental variables using one fishing occasion at each site (n = 6104) from FIsh Database of European Streams

	PC1 'Latitude'	PC2 'Size'	PC3 'Altitude'
Mean air temperature	−0.90		
Latitude	0.81		
Longitude	0.77		
Distance from source		0.90	
Size of catchment class		0.81	
Wetted width		0.78	
Altitude			0.88
Gradient slope			0.66
Conductivity class			−0.64
Initial eigenvalues	2.6	2.2	1.4
Percent of variance explained	29.3	24.0	15.4
Cumulative percent	29.3	53.3	68.7
Extraction method: principal component analysis			
Rotation method: varimax with Kaiser normalisation (four iterations)			

If fishing occasions had been used as calibration data for the European Fish Index (Pont *et al.* 2007) that fishing occasion was used, otherwise one fishing occasion was randomly selected for each site. Only loadings >0.5 are shown.

Table 5. Spearman rank correlation coefficients of fish metrics and principal component scores from FIsh Database of European Streams based on environmental variables, using one fishing occasion at each site (n = 6104)

	PC1 'Latitude'	PC2 'Size'	PC3 'Altitude'
Intolerant species (relative number)	0.41		0.61
Tolerant species (relative number)	−0.25		−0.54
Benthic species (number of)		0.24	−0.24
Rheophilic species (number of)	0.20	0.23	
Lithophilic density (relative ind. ha^{-1})	0.30		0.67
Phytophilic density (ind. ha^{-1})			−0.30
Insectivore density (ind. ha^{-1})			0.58
Omnivore density (ind. ha^{-1})	−0.21		−0.39
Long migrating species (number)			−0.33
Potamodromous species (number)		0.47	

Only coefficients >0.2 are shown.

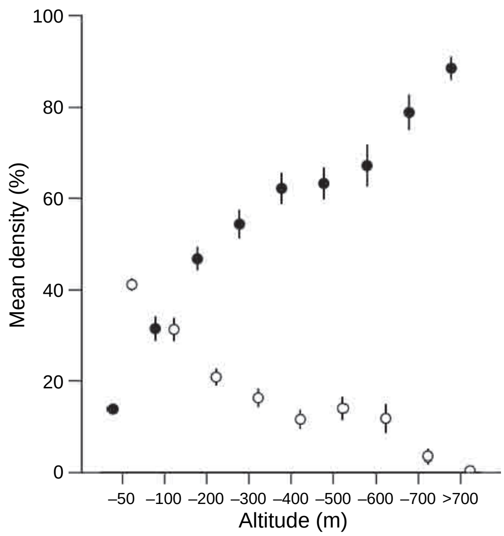

Figure 2. Mean relative densities (%) of the fish feeding guilds insectivores (●) and omnivores (○) ±95% CI plotted against altitude class of fishing site in FIsh Database of European Streams, representing 8228 fishing occasions at sites. If the fishing occasion at the site was used as calibration data for the European Fish Index (Pont *et al.* 2007) that fishing occasion was used, otherwise one fishing occasion was randomly selected for each site.

General relationships between environment and fish community

FIsh Database of European Streams covers a wide geographic and climatic range within Europe (Table 3). The variables mean air temperature, latitude, longitude, gradient (CEN 2003), distance from source, size of catchment class (Oberdorff, Pont, Hugueny & Chessel 2002), wetted width, altitude (Oberdorff *et al.* 2002; CEN 2003) and conductivity class (Hornung & Reynolds 1995) were simplified using principal component analysis (PCA). The PCA of selected environmental variable revealed three main gradients throughout Europe; PCA axis 1 indicated that both latitude, longitude and air temperature separated sites, followed by variables that measure stream size (PCA2) and then altitude/slope, which was weakly inversely correlated with conductivity (Table 4). These main environmental factors were each correlated with the fish fauna guilds (Table 5). In the north, intolerant, rheophilic and lithophilic species were more common, and intolerant as well as omnivorous species were less predominant. In large rivers, benthic, rheophilic and potamodromous species were common. At higher altitudes, intolerant, lithophilic and insectivorous species were more common, resembling the situation in northern rivers, while tolerant, benthic, phytophilic, omnivorous and long migrating species were less common (Table 5). There was a strong and consistent pattern between the percentage insectivores and omnivores with altitude class (Fig. 2). Similarly, rheophilic species were less common in larger, lowland rivers. In contrast, benthic species were more common in larger streams, while limnophilic species seem less affected by stream size (Fig. 3).

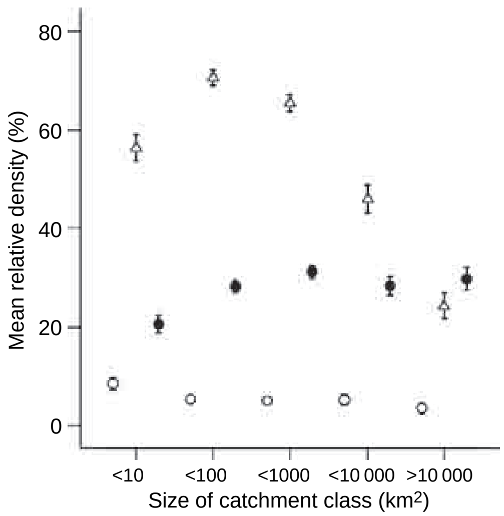

Figure 3. Mean relative densities (%) of the fish habitat guilds limnophilic species (white circles), benthic (black circles) and rheophilic (triangles) ±95% CI plotted against size of catchment class of fishing site in FIsh Database of European Streams, representing 8228 fishing occasions at sites. If the fishing occasion at the site was used as calibration data for the European Fish Index (Pont *et al.* 2007) that fishing occasion was used, otherwise one fishing occasion was randomly selected for each site.

Discussion

FIsh Database of European Streams has a wide geographic and environmental representation within Europe and contains quantitative data on a considerable number of fish species as well as human pressures (Degerman, Beier, Breine, Melcher, Quataert, Rogers, Roset and Simoens 2007). This information has been used to develop the European Fish Index (Pont *et al.* 2007). It is a potential source of information for many other purposes including testing fundamental ecological theories as well as applied research concerning environmental protection and management of resources. For example, the database contains detailed information on methods and sampling strategies that could be used to evaluate costs and benefits for monitoring programmes, or aid a future revision of electric fishing standards (CEN 2003). The data also have applications in large-scale phylogeographical, ecological and conservation studies. On a European scale, the distribution of threatened and endangered species can be linked to potential damaging impacts between rivers and regions.

There are several other advantages in merging data gathered locally. First, the data have cost time, money as well as effort to collect, and to make full use of such a resource makes sense with research income becoming increasingly difficult to obtain. Secondly, analyses that are more complex are possible on such a large data set with such a wide geographical coverage, thus extending knowledge on meta-population problems.

An important conclusion from the derived relationships between environmental variables on one hand, and their respective correlations with fish guilds on the other, is that the fish guilds reflect the gradients in environmental variation (Figs 2 and 3). Using guilds instead of species, and also relating biological and environmental gradients strengthened the approach of the European Fish Index (Pont *et al.* 2007). Furthermore, the overview of environmental data shows that there was no geographical or climatic bias in the two main environmental factors – river size and altitude.

Developing FIDES provided the impetus for standardisation of concepts, methods and variables to be used across the EU. This process forces parties to evaluate priorities and agree on definitions, and raises crucial questions that need answering before continuing. The construction of a database develops a feedback process between researchers, end-users and managers, as authorities who provide the data most often have to implement the findings of the research. Finally, the process of standardisation provides a higher level mechanism for quality assurance and quality control, thus making each national database more valuable.

The development of technology and international co-operation provide the opportunity for large and open access databases, but there are problems in achieving this vision. Strong traditions and circumstances prevent openness towards data sharing. Biologists tend to work in a market where governmental funding finances research and professional positions. Private consultants are a threat to universities and governmental institutes, as commercial businesses would benefit greatly from open data sources, and mean even more competition for funding. It is, therefore, understandable that scientists who produce data are often unwilling to share it. Furthermore, to make knowledge available of the occurrence of sensitive species involves a certain risk of, for example, uncontrolled exploitation. Consequently, mechanisms are needed to standardise and make data available at the global level if science and the environment are to benefit from the utility of databases.

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References

Angermeier P.L. & Karr J.R. (1986) Applying an index of biotic integrity based on stream fish communities: considerations in sampling and interpretation. *North American Journal of Fisheries Management* **7,** 331–338.

AQEM (2002) *The Development and Testing of an Integrated Assessment System for the Ecological Quality of Streams and Rivers Throughout Europe Using Benthic Macroinvertebrates*. EU-Project Contract No. EVK1-CT1999-00027. Available at: http://www.aqem.de. [Last accessed on 9 October 2007.]

Bohlin T., Hamrin S., Heggberget T.G., Rasmussen G. & Saltveit S.J. (1989) Electrofishing – theory and practice with special emphasis on salmonids. *Hydrobiologia* **173,** 9–43.

CEN (2003) *Water Quality – Sampling of Fish With electricity*. . European Standard – EN 14011:2003. Brussels: European Committee for Standardization, 18 pp.

Cowx I.G. (ed.) (1990) *Developments in Electric Fishing*. Oxford: Fishing News Books, 368 pp.

Cowx I.G. (1995) Fish stock assessment – biological basis for sound ecological management. In: D. Harper & A. Ferguson (eds) *Biological Basis for River Management*. London: Wiley and Son, pp. 375–388.

Cowx I.G. & Lamarque P. (eds) (1990) *Fishing with Electricity*. Applications in Freshwater Fisheries Management. Oxford: Fishing News Books, 248 pp.

Degerman E., Beier U., Breine J., Melcher A., Quataert P., Rogers C., Roset N. & Simoens I. (2007) Classification and assessment of degradation in European running waters. *Fisheries Management and Ecology* **14,** 417–426.

Dekker W. (2003) Did lack of spawners cause the collapse of the European eel, *Anguilla anguilla*? *Fisheries Management and Ecology* **10,** 365–376.

Hering D., Moog O., Sandin L. & Verdonschot P.F.M. (2004) Overview and application of the AQEM assessment system. *Hydrobiologia* **516,** 1–20.

Hernandez M.J. (2003) *Database Design for Mere Mortals: A Hands-On Guide to Relational Database Design*, 2nd edn. Boston: Addison-Wesley Professional, 611 pp.

Hornung M. & Reynolds B. (1995) The effects of natural and anthropogenic environmental changes on ecosystem processes at the catchment scale. *Trends in Ecology and Evolution* **10,** 443–449.

ICES (2004) *International Council for the Exploration of the Sea*. Available at: http://www.ices.dk.

Jones M.L. & Stockwell J.D. (1995) A rapid assessment for the enumeration of salmonid populations in streams. *North American Journal of Fisheries Management* **16,** 551–562.

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

Karr J.R. & Chu E.W. (1999). *Restoring Life in Running Waters: Better Biological Monitoring*. Washington, DC: Island Press, 206 pp.

Melcher A., Schmutz S., Haidvogl G. & Moder K. (2007) Spatially based methods to assess the ecological status of European fish assemblage types. *Fisheries Management and Ecology* **14,** 453–463.

Noble R.A.A., Cowx I.G., Goffaux D. & Kestemont P. (2007) Assessing the health of European rivers using functional ecological guilds of fish communities: standardising species classification and approaches to metric selection. *Fisheries Management and Ecology* **14,** 381–392.

Oberdorff T., Pont D., Hugueny B. & Chessel D. (2001) A probabilistic model characterizing fish assemblages of French rivers: a framework for environmental assessment. *Freshwater Biology* **46,** 399–415.

Oberdorff T., Pont D., Hugueny B. & Porcher J.P. (2002) Development and validation of a fish-based index (FBI) for the assessment of "river health" in France. *Freshwater Biology* **47,** 1720–1734.

Pont D., Hugueny B. & Rogers C. (2007) Development of a fish-based index for the assessment of river health in Europe: the European Fish Index. *Fisheries Management and Ecology* **14,** 427–439.

Roset N., Grenouillet G., Goffaux D., Pont D. & Kestemont P. (2007) A review of existing fish assemblage indicators and methodologies. *Fisheries Management and Ecology* **14,** 393–405.

Zippin C. (1958) The removal method of population estimation. *Journal of Wildlife Management* **22,** 82–90.

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Spatially based methods to assess the ecological status of riverine fish assemblages in European ecoregions

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Abstract The objective was to develop spatially based (type-specific) methods to assess the ecological status of European rivers according to the EU Water Framework Directive. Some 15 000 samples from about 8000 sites were pre-classified within a five-tiered classification system based on hydromorphological and physico-chemical pressures. The pre-classification was used to identify reference conditions and to calibrate the assessment methods. Clustering reference sites based on relative species composition resulted in 60 fish assemblage types within 11 of the ecoregions under study. Discriminant function analyses (DFAs) were employed to identify environmental parameters characterising fish assemblage types; altitude, river slope, wetted width, mean air temperature and distance from source were the principal predictors. These environmental parameters were used to assign impacted sites with altered fish assemblage composition to the reference fish assemblage type. Metrics (fish assemblage descriptors) responding to human pressures were selected based on correlation and DFAs. Assessment methods were developed for 43 fish assemblage types. Metrics based on individual sentinel species were more often used in type-specific methods than metrics related to reproduction, habitat and feeding. Metrics based on long-distance migrants and potamodromous species were more sensitive to human pressures than overall composition metrics, e.g. total number of species. Only some of the tested metrics showed pressure-specific responses, i.e. reacted to one type of pressure but not to others. Insectivorous, intolerant and lithophilic species exclusively responded (decreased) to chemical and hydromorphological pressures in 14–19%. Omnivorous species was the only metric type that showed a consistent reaction (increase) to continuum disruptions in 25% of the cases. Accuracy of methods based on cross-validation with pre-classification varied between 47% and 98% (mean 81%) when contrasting calibration data set (class 1 and 2) with degraded sites (class 3, 4 and 5).

KEYWORDS: ecological status assessment, fish, index of biotic integrity, methods, Water Framework Directive.

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Introduction

The development of fish-based methods to assess human-induced impacts on aquatic ecosystems has been strongly influenced by the index of biotic integrity (IBI; Karr 1981). IBIs are based on the assumption that within a given entity the variability of fish communities is low enough to be able to distinguish between natural and human-induced variability. Although IBIs have been developed worldwide (Roset, Grenouillet, Goffaux, Pont & Kestemont 2007), appropriate delineation of regions or river types is a major issue in developing regional IBIs (Strange 1998) because of large-scale natural variability in fish communities. IBIs developed in North America account for natural variability by regionalisation and metric adjustments to longitudinal gradients (e.g. Hughes, Howlin & Kaufmann 2004).

The spatially based approach is the underlying methodological principle of the European Union's Water Framework Directive (WFD; 2000/60/EEC) for assessing the ecological status of running waters. The concept is that rivers can be classified into units (e.g. river segments) with homogenous characteristics. The WFD offers two options: System A and System B; both use only abiotic criteria for river typology. However, as the hypothesis is that the less the biotic heterogeneity within identified types the higher the accuracy of the employed IBI (Fausch, Lyons, Karr & Angermeier 1990; Smogor & Angermeier 1998, 2001), biotic typologies should be used. In principle, two spatial dimensions structure fish assemblages at the large scale: (1) the zoogeographic pattern reflecting mainly climatic gradients across the continental scale and (2) the longitudinal pattern within each river at the catchment scale caused by changes in environmental parameters along the river course (Beier, Degerman, Melcher, Rogers & Wirlöf 2007). Ecoregions are supposed to provide a spatial framework for ecosystem assessment at the large scale (Omernik 1987, 1995). Illies (1967) introduced a European classification system, dividing the continent into 25 ecoregions. Although Illies' ecoregion system is the only widely used classification, and has been adopted by the WFD, it has never been evaluated for its ability to discriminate among fish communities at a continental scale (Reyjol, Hugueny, Pont, Bianco, Beier, Caiola, Casals, Cowx, Economou, Ferreira, Haidvogl, Noble, de Sostoa, Vigneron & Verbickas 2007). The longitudinal zonation of rivers with a sequence of distinct fish communities was developed more than a century ago (Fritsch 1872). Several key parameters, e.g. slope/width ratio (Huet 1949), have been used to explain longitudinal fish community patterns in specific regions, but no attempt has been made to analyse longitudinal patterns at the ecoregion and continental scale. In the WFD, the classification of surface water bodies is based on the assumption that an abiotic river typology is adequate to stratify fish communities, but no efforts to validate this assumption at a European scale have been undertaken.

According to the IBI concept, assessment of ecological status is based on the comparison between observed and expected values of a set of defined metrics (fish assemblage descriptors) (Noble, Cowx, Goffaux & Kestemont 2007) combined into a single index (Karr 1981). Expected conditions can be defined in different ways (Hughes 1995), one is to use best available (least-disturbed) conditions resulting in unequal thresholds for less and more impacted fish assemblage types. However, the WFD requires standardised reference conditions showing no, or only minor, anthropogenic alterations. Therefore, most of the existing methods are not compliant with the WFD.

Information on pressures is necessary to distinguish between reference and impacted sites and for calibrating or scoring of metrics. However, compared with other aspects of IBI development, less emphasis has been dedicated to quantifying precisely the level of degradation.

Spatially based assessment methods (SBM) were developed for the WFD based on data from ecoregion 1 (Iberian Peninsula), 2 (Pyrenees) 4 (Alps) 8 & 13 (Western Highlands & Western Plains), 9 (Central Highlands), 14 (Central Plains), 15 (Baltic Province), 18 (Great Britain) 20 & 22 (Borealic Upland & Fenno-Scandinavian Shield). Detailed descriptions of ecoregional SBM developments are found in project reports (http://fame.boku.ac.at) and in examples presented in this volume on ecoregion 1 (Ferreira *et al.* 2007), ecoregions 8 & 13 (Grenouillet *et al.* 2007), ecoregion 15 (Virbickas & Kesminas 2007) and ecoregion 18 (Noble, Cowx & Starkie 2007). The objective of this paper was to summarise and to compare methodologies and results in these 11 ecoregions (Fig. 1) with respect to: (1) standardised identification and characterisation of fish assemblage types; (2) abiotic characteristics which discriminate among fish assemblage types; (3) metrics response to human pressures; (4) metrics aggregation and status classification; and (5) accuracy of assessment methods.

Figure 1. Illies' (1967) ecoregions. Ecoregions analysed in this paper are in bold.

Methods

All data were extracted from an extensive database (FIDES; Beier *et al.* 2007). For the spatially based approach, data from 14 789 fishing occasions at 7977 sites collected in 11 ecoregions and 11 countries were available. Fisheries data were based on single-pass electric fishing methodology only (Beier *et al.* 2007). Species were classified as alien or native (Noble *et al.* 2007).

Pre-classifications of sites based on potential pressures (Degerman, Beier, Breine, Melcher, Quataert, Rogers, Roset & Simoens 2007) were used to select unimpacted or weakly impacted sites for fish assemblage typology development and to calibrate the developed assessment methods against a five-tiered pressure gradient. The five classes of pressure status were defined following the normative classification of biological quality elements of the WFD: a pressure level that is supposed to result in minor impact on fish is equivalent to class 1, slight = class 2, moderate = class 3, strong = class 4 and severe = class 5.

Fish-based river typologies (i.e. fish assemblage types) for ecoregions were developed by cluster analyses and other techniques (see results) using only unimpacted or weakly impacted sites. To allocate new (impacted) monitoring sites to fish assemblage types (hereafter fish types) several environmental descriptors (Beier *et al.* 2007) were tested for their power to discriminate among fish types using stepwise discriminant analysis (SPSS 12.0®). Discriminant function analysis (DFA) generates functions with linear combinations of variables that maximises the probability of correctly assigning observations to their pre-determined groups. DFA can also be used to classify new observations into one of the groups (Quinn & Keough 2002). DFA has been used for the same and similar purposes in RIVPACS (Moss 2000), a system developed for invertebrates in the UK, and for predictive fish models (Joy & Death 2002).

More than 400 potential metrics (Noble *et al.* 2007) were analysed for their capacity to respond to human pressures. Candidate metrics, e.g. metrics with clear dose–response relationship, were tested for redundancy. Pressure-specific response of metrics was tested as follows. A metric was classified as pressure specific if it responded (Spearman's $r > |0.6|$) within a fish type to one type of pressure only, i.e. chemical (mean of nutrients/organic input and toxics/acidification/O_2), physical (mean of morphology and hydrology) or connectivity but not to another pressure type. Different options to aggregate final metrics and to set class boundaries among ecological status classes were evaluated (e.g. box plots and multivariate statistics). Error of spatially based methods was assessed by cross-validation between pre-classification of pressures and ecological status classification. Accuracy differences were tested among ecoregions and fish zones by ANOVA using the robust Welch-statistic for heterogeneous variance (SPSS 12.0®). Huet's classification (Huet 1949) was used to analyse differences along the longitudinal zonation.

Results

Data available for the analyses were originally collected for different purposes (e.g. fish species inventory, impact assessment studies and conservation monitoring; Beier *et al.* 2007). Hence, available fish data were unevenly distributed across Europe and countries and covered mainly western (Northern Portugal, France, England, Belgium and the Netherlands) and central Europe (Austria) (Beier *et al.* 2007; Reyjol *et al.* 2007).

Reference conditions and pressure status

Five core pressure variables (morphology, hydrology, nutrients/organic input, toxics/acidification/O_2 and connectivity) were combined to a mean global pressure index and used to select weakly impacted sites in all ecoregions. Other pressure variables (land use, urbanisation, riparian zone, sediment load, upstream dam, salinity, impact of stocking and introduction of fish) were added to account for regional peculiarities, where data were available (Table 1). In general, weakly impacted sites (calibration sites) encompass only pressure status classes 1 and 2 (minor and slight impact expected). However, because of the low number of non-impacted sites in ecoregions 1, 14, 15 and 18 some pressure variables also included sites with

Table 1. Pressure variables used to distinguish between calibration and impacted data set in different ecoregions (for definition of pressure variables see Degerman *et al.* 2007)

Human pressures	Iberian Peninsula 1	Pyrenees 2	Alps 4	Western Highlands & Western Plains 8 & 13	Central Highlands 9	Central Plains 14	Baltic Province 15	Great Britain 18	Borealic Upland & Fenno-Scandian Shield 20 & 22
Morphology	×	×	×	×	×	×	×	×	×
Hydrology	×	×	×	×	×	×	×	×	×
Toxics/acidification/O_2	×	×	×	×	×	×	×	×	×
Nutrients/organic input	×	×	×	×	×	×	×	×	×
Connectivity_segment	×	×		×	×		×	×	×
Land_use	×								
Urbanisation	×								
Riparian_zone	×								
Sediment_load	×								
Upstream_dam	×								
Salinity	×								
Impact_of_stocking									×
Introduction_of_fish									×

pressure status class 3, resulting in a contribution of moderately impacted sites of 28, 54, 26 and 39 percent in these four ecoregions. In total, 1920 calibration sites were separated from 4554 'impacted sites', encompassing 2200 rivers and 102 fish species.

Fish types

Ecoregions 20 and 22 as well as 8 and 13 were combined as preliminary analyses did not reveal significant differences in fish-type patterns based on cluster analyses. As a rule, relative abundance (catch data of the first run) and hierarchical cluster analyses (Ward's method) were used to group calibration samples with similar species composition except in ER2 and ER8/13 (absolute abundance), ER18 (absolute abundance standardised per species) and ER 20/22 (expert judgement process) (Table 2). In total, 60 fish types were identified ranging from two (Pyrenees) to eight types per ecoregion (median: 6.6 types per ecoregion). Eight types were added based on historical data or expert judgement but are not included in further analyses. Following Huet's classification, the highest proportion of types (31 types) corresponded to the trout, 13 to the grayling, 14 to the barbel and two to the bream zone. The mean number of fish species per type and sample ranged from 0.9 to 17.2 (median 5) with the lowest values in the trout region (0.9–5.9). *Salmo trutta fario* L. was the dominating species in 20 types followed by *Phoxinus phoxinus* (L.) (10 types), *Rutilus rutilus* (L.) and *Salmo salar* L. (four types each). When only the five most abundant species per fish type were considered, the total diversity comprised only 47 species.

Discriminating fish types

In general, DFAs were used to identify environmental descriptors predicting fish types and assigning degraded sites to fish types. In ecoregion 20/22, an expert decision system was employed (Table 2). Correct assignment of sites to fish types using environmental descriptors ranged from 18% (ER 14) to 100% (ER 1). Between 2 and 6 environmental descriptors (median 5) were used to discriminate fish types. Altitude, river slope, wetted width, mean air temperature (annual and/or July) and distance from source were the prevailing environmental descriptors.

Metrics response

In all ecoregions, the mean of the five human pressure variables was used as an index for human pressure. Spearman's rank correlation (nine ecoregions) and discriminant analysis (four ecoregions) were the primary methods to analyse metrics response to human

Table 2. Methods used for spatially based assessment method development

Method	Iberian Peninsula 1	Pyrenees 2	Alps 4	Western Highlands & Western Plains 8 & 13	Central Highlands 9	Central Plains 14	Baltic Province 15	Great Britain 18	Borealic Upland & Fenno-Scandian Shield 20 & 22
Fish types									
Cluster analyses	×	×	×	×	×	×	×	×	–
Expert judgement	–	–	–	–	–	–	–	–	×
Discrimination between types									
Discriminant analyses	×	×	×	×	×	×	×	×	–
Selection of metrics									
Spearman rank correlation	×	×	×	×	×	×	×	×	–
Test of metrics redundancy									
Spearman rank correlation	×	×	×	–	×	–	×	×	–
Class boundaries									
Discriminant analyses	–	–	×	×	×	–	–	×	–
Box plots	×	×	–	–	–	×	×	–	–
Mann-Whitney *U*-test	×	–	–	–	–	–	–	–	–
Kruskal–Wallis	–	×	–	–	–	–	–	–	–
Metrics combination									
Discriminant analyses	–	–	×	×	×	–	–	×	–
Multimetric index	×	×	–	–	×	×	×	–	–
Method validation									
Independent data set	–	–	–	×	–	–	–	×	–

pressures (Table 1). For a more detailed description of how discriminant analysis was used for ecological status allocation see Melcher, Schmutz, Haidvogl & Moder (2007).

Only responsive metrics (Spearman's rank correlation $P < 0.05$ or $r > |0.4|$) were selected for further analyses. Redundancy among metrics was tested by Spearman's rank correlation and metrics with $r > |0.80|$ were excluded. In ecoregions 1, 2 and 14, each metric was scored from 1 to 5 by plotting metric values against mean pressure (box plots). The final index was calculated as the average metric score. In ecoregion 9, metrics were normalised from 0 to 1 and the average of the metrics was scored by box plots (index vs mean pressure). Scores and class boundaries were set either for each single metric (ecoregions 9, 14 and 15) or by discriminant analysis (ecoregions 4, 8/13, 15 and 18), as it was done for metrics selection using the pre-classification of pressures for calibration (Table 1).

Spatially based methods were developed for 43 fish types in eight ecoregions (total number of samples: 10 120). The number of available samples per fish type ranged from 16 to 1120 (median 235). For the remaining fish types ($n = 16$) the number of samples or their distribution along pressure gradient was not adequate for method development. All fish types in ER 20/22 were omitted because of lack of sufficient impacted sites. The Swedish database used for this approach covered only small streams that are less likely to be impacted.

In total, 130 different metrics (of 451 metrics tested) were selected with a median of 9.2 metrics per fish type, with percentage of insectivorous individuals, percentage of lithophilic individuals and percentage of lithophilic species selected most often. The 10 most often used metrics represent about one-third of selected metrics. All of the 10 most often used metrics represent relative measurements (nine of 10) or presence/absence information (Fig. 2).

Similar metrics, based on grouping metrics into different types (i.e. functional metrics and sentinel metrics) and variants (i.e. same units) were used for all fish zones, i.e. trout, grayling and barbel zone (in the bream region only two fish types were identified). Metrics based on individual sentinel species were most commonly selected followed by metrics related to reproduction, habitat and feeding. About one-third of all sentinel metrics were based on *S. trutta fario*. Metrics based on long-distance migrants and potamodromous species were more important than overall composition metrics (e.g. total number of species) (Table 3). With respect to metric variants, density metrics (ind ha^{-1} and % ind ha^{-1}) were used as often as all other metrics such as number of species, biomass or 0+ fish (Table 4).

Only some of the tested metrics showed pressure-specific response, i.e. reacted to one type of pressure but not to another. Insectivorous, intolerant and lithophilic metrics exclusively responded (decreased) to chemical and hydromorphological pressures in only 14–39% of the cases. However, in some cases, these metrics showed no or an opposite response (increase) to connectivity disruptions. Only omnivorous species (metrics) showed a consistent reaction (increase) to disruptions of the continuum, in 25% of the cases (Table 5).

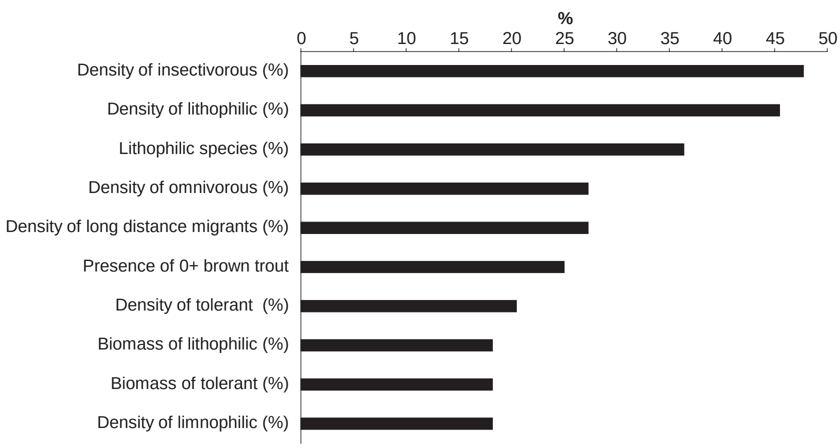

Figure 2. Ten metrics most often used within 43 fish type-specific methods of eight ecoregions (calculated as the percentage of use within the 43 fish types, (%) represent relative measurements).

Table 3. Relative importance of metric types (% of all metrics used) in different fish zones

	Total	Trout	Grayling	Barbel	Bream
Number of types	44	31	13	14	2
Sentinel	23.4	18.4	21.2	31.8	5.0
Reproduction	17.6	17.1	18.2	15.6	35.0
Habitat	15.1	17.7	21.2	10.4	10.0
Feeding	14.9	15.2	16.7	12.3	25.0
Tolerance	10.1	7.6	7.6	12.3	20.0
Migration	7.3	6.3	12.1	6.5	5.0
Overall composition	6.0	9.5	1.5	5.2	0.0
Longevity	2.8	3.2	0.0	3.9	0.0
Historical metrics	1.5	3.8	0.0	0.0	0.0
0+	1.3	1.3	1.5	1.3	0.0

Sentinel: based on individual sentinel species; overall composition: species diversity, native species, exotic species, etc.; historical metrics: based on absence/presence of historically documented species; 0+: occurrence, number or relative number of YOY individuals of sentinel species; for further details on metrics definition see Noble *et al.* (2007).

Table 4. Relative importance of metric variations (% of all metrics used) in different fish zones

	Total	Trout	Grayling	Barbel	Bream
No. of fish types	44	31	13	14	2
% density	31.2	20.9	45.5	32.0	60.0
Density	20.4	17.7	15.2	28.1	0.0
% species	16.9	20.9	13.6	13.7	20.0
Biomass	9.6	9.5	10.6	10.5	0.0
No. of species	9.6	16.5	6.1	5.2	0.0
% biomass	6.8	6.3	3.0	7.2	20.0
Presence 0+	4.0	5.7	4.5	2.6	0.0
% presence 0+	1.5	2.5	1.5	0.7	0.0

% density/% biomass: relative density/biomass; density/biomass: number of individuals per hectare; % species: proportion of species compared with all species; (%) presence 0+: (%) number of sentinel species with YOY fish; all metrics are based on single-pass electric fishing samples; for further details on metrics definition see Noble *et al.* (2007).

Method accuracy

Accuracy of methods based on cross-validation with pre-classification varied between 47% and 98% (mean 81%) when contrasting calibration data set (class 1 and 2) with degraded sites (classes 3, 4 and 5). Method accuracy was different among ecoregions (Welch test = 7.914, d.f. = 7, P = 0.002) but homogeneous across fish zones (Welch test = 0.018, d.f. = 3, P = 0.996). The lowest accuracy was achieved in the Alps ecoregion (58%), and the highest in Western Highlands/Western Plains (87%). Mean type I error in classification, i.e. a site is classified by the method as impacted, although the pre-classification indicated no severe pressures, was 8.9% (range 0–47%) and similar to type II error, i.e. a site is classified by the method as unimpacted although the pre-classification indicates severe pressures, ranging from 0% to 33% (mean 10.2 %). Sample size had a significant, but marginal, influence on the method accuracy (r^2 = 0.13, F = 7.4, P < 0.01). Methods were validated by independent data sets in only two ecoregions (Table 2).

Table 5. Pressure-specific response of functional metrics calculated as exclusive response (Spearman's $r_s > |0.6|$) to one of the pressure groups (in percent); − indicates a decrease; + an increase of the metrics (only metrics that show pressure-specific response in at least 25% of the 43 methods in one category are shown; sentinel species metrics are not considered)

	Pressure					
	Chemical		Hydro-morpho-logical		Connectivity	
Metric	−	+	−	+	−	+
Insectivorous	39	0	20	0	0	9
Intolerant	39	0	16	0	14	5
Lithophilic	25	0	14	0	0	2
Rheophilic	27	0	9	0	0	0
Omnivorous	0	7	0	7	0	25

Discussion

This paper represents one of the first attempts to develop data-driven assessment methods for running waters at the European scale. Based on a large European database (FIDES), sampling sites were pre-classified according to human pressures to distinguish between calibration and impacted sites. Calibration sites were used to develop fish typologies of running waters within 11 ecoregions. Environmental descriptors were identified for discriminating among fish types and for allocating impacted sites to fish types. Type-specific metrics responding to pressures were used to discriminate between ecological status classes allowing the allocation of new sites to ecological status classes (Fig. 3).

Fish types

The data covered a wide range of different ecoregions and rivers types representative of western, central and northern Europe. However, there was a paucity of data for Mediterranean and eastern countries.

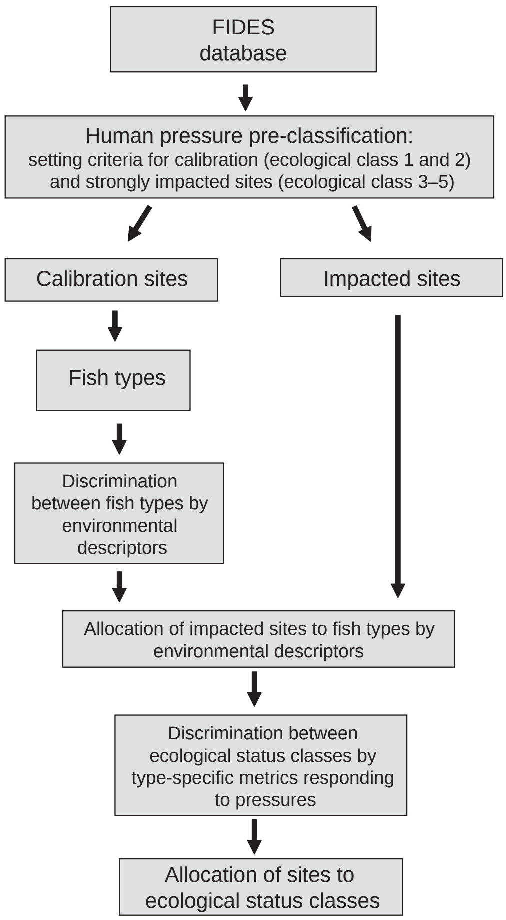

Figure 3. Flow chart for the development of spatially based assessment methods.

One of the major challenges of this pan-European approach was the standardised selection of reference sites according to the WFD criteria. Strictly following the WFD criteria resulted in a lack of reference sites in some ecoregions (ecoregions 1, 14, 15 and 18) and sites with pressure status 3 had to be included for the development of the fish types. Such an approach seems not to violate the WFD rules as calibration data were only used for fish type identification but not for discriminating among the different levels of deterioration within the final assessment methods. However, there still remains the question of how much deterioration at calibration sites affects fish typology.

Most IBIs were developed for specific regions or countries. Within-region variability of reference sites has received little attention. In the proposed approach, reference sites were clustered using relative species composition into fish types and two to eight distinct fish types were identified in each of the ecoregions. As the level of discrimination among groups within cluster analyses is arbitrary and relies on expert judgement, other thresholds would have resulted in different groupings. However, there is a trade-off between defining groups as small and hence homogeneous as possible and the availability of data. As data availability varied among ecoregions and some types by nature are scarce the criteria for defining thresholds in the examination of the cluster dendograms also differed among ecoregions. It was assumed that these differences did not affect the accuracy of assessment methods developed as Melcher *et al.* (2007) showed more broadly defined fish assemblage types produced reliable results.

The spatio-temporal pattern of fish communities is a result of a plethora of factors acting across several scales. Both top-down factors, e.g. zoogeography, geomorphology, and bottom-up factors, e.g. habitat availability and competition, are known to determine the structure of fish assemblages. Smogor & Angermeier (1998) used physiographical regions for delineating IBI regions to account for upland and lowland river section differences. To account for within-region variability, Fausch, Karr & Yant (1984) suggested using stream order or catchment area. Some authors have tested stream-size dependence of IBI scores or metrics (e.g. Mundahl & Simon 1998) or adjusted metrics to watershed size (e.g. Hughes *et al.* 2004). However, in most IBI developments no adjustments have been made (Smogor & Angermeier 1998). The DFAs showed that not only stream size (wetted width), but also altitude, slope, mean air temperature and distance from source have effects on species composition, parameters that correlate only partly with stream size. A median number of five environmental descriptors were used to discriminate among fish types. Most identified environmental parameters structuring fish assemblages describe the longitudinal gradient of streams and rivers.

In the USA, separate IBIs have been developed for cold (e.g. Hughes *et al.* 2004) and warm water streams (e.g. Karr 1981). The present results, however, show that fish communities not only differ considerably both among biogeographical regions and among cold and warm waters but also within cold and warm waters as a result of the longitudinal zonation. For example, in the Alps ecoregion, six different fish types were found for cold water streams (trout and grayling fish zone) showing different proportions of brown trout or even dominance of cyprinids (*Leuciscus souffia* Risso). However, developing a high number of spatially homogeneous units accounting mainly for longitudinal gradients resulted in a deficit of number of sites left per individual fish type preventing IBI development in some of the types.

The diversity of fish fauna in European rivers is low compared with North America. The mean expectation of number of fish species based on the database was four, showing an increase from the trout to bream zones. Forty-seven species accounted for the five most abundant species expected at a given site in all the fish types identified. Although Mediterranean rivers are under-represented in the database, about one-quarter of all species are endemic to the Mediterranean region underlining the distinctness of southern European fish fauna and by comparison, the homogeneous fish communities in western, central and northern Europe.

Human pressure pre-classification

Hydromorphological alterations, water pollution and continuum disruptions combined into a mean pressure index, functioned as a calibration index for metric selection and class boundary setting. Unlike other IBI developments, a five-tiered pre-classification system was used. This had several advantages: (1) identification of fish types where insufficient numbers of reference sites or impacted sites with different levels of degradation were available – these could be eliminated before proceeding with IBI development; (2) pre-classification was used to distinguish between reference and impacted sites and to calibrate the fish-based index; and (3) comparing IBI scores with pressure pre-classification enabled cross-validation of the methods developed.

Metrics

Although more than 400 metrics were tested, few were eventually employed. The most commonly selected metrics were percentage of insectivorous individuals, percentage of lithophilous individuals and percentage of lithophilous species. Insectivorous (alternately invertivorous; Oberdorff, Pont, Hugueny & Chessel 2001; Belpaire, Smolders, Vanden Auweele, Ercken, Breine, Van Thuyne & Ollevier 2000; Breine, Simoens, Goethals, Quataert, Ercken, Van Liefferinghe & Belpaire 2004) and lithophilous (Oberdorff & Hughes 1992) were used in other European IBIs.

Grouping metrics into metric types and exploring their relative importance showed that sentinel species were the predominant type of metric used in the 43 methods (Table 3). The spatially based approach minimised natural variability within fish types. This favoured the selection of species-specific (sentinel) metrics over other metrics, as the clustering of fish types was based on relative species composition. Species were classified as sentinel species if they were typical of the distinct fish communities (e.g. fish zones), sensitive to human disturbances and sufficiently abundant and well distributed under undisturbed conditions (Noble *et al.* 2007). About one-third of all sentinel metrics were based on brown trout, indicating that in species poor fish types sentinel species metrics were more important than functional metrics (Hughes *et al.* 2004).

In addition to species-specific metrics, functional metrics referring to reproduction, habitat and feeding were used in all types and fish zones. Newly introduced metrics such as long-distance migrants and potamodromous species were less often used, but are more frequently employed (7.1%) than metrics such as total number of species (6.1%) that were used in earlier IBIs (Karr 1981). This might arise because species diversity can react to human pressures both in terms of a decrease (e.g. loss of intolerant species) and an increase (e.g. additional tolerant species). An increase of species number was observed, for example, in impounded rivers (Martinez, Chart, Trammell, Wullschleger & Bergersen 1994). Historical metrics (e.g. % of original species) were rarely used because few countries (Austria, Germany and the Netherlands) provided full data sets. However, historical metrics are the only option to develop assessment methods in regions and fish types, e.g. ecoregions 14 and large rivers, where appropriate reference sites are no longer available (de Leeuw, Buijse, Haidvogl, Lapinska, Noble, Repecka, Virbickas, Wisniewolski & Wolter 2007).

With respect to metric variants, density metrics (ind ha^{-1} and % ind ha^{-1}) were used as often as all other metric variants combined, i.e. number of species, biomass and 0+ fish. This confirms the general trend in IBI developments that at least semi-quantitative information on fish communities is necessary to detect human impacts. Biomass and 0+ data were supplied by only a few countries; therefore, those metrics were under-represented in the database, but proved to be effective where data were available (e.g. ecoregion Alps and fish-type hyporhithral, see Melcher *et al.* 2007).

Only some of the tested metrics showed pressure-specific response, i.e. reacted to one type of pressure but not to another. Insectivorous, intolerant and lithophilic species decreased exclusively to chemical and hydromorphological pressures in 9–39% of the cases. However, in some cases, those species show no or an opposite response (increase) to connectivity disruptions. Omnivore metrics were the only ones that showed a consistent reaction (increase) to continuum disruptions in 25% of the cases.

After identifying impacted sites, the next step of water management is to identify appropriate restoration measures. For that, the index or individual metrics should be able to infer which type of restoration measures would be appropriate. More detailed investigations on pressure-specific response of fish are necessary to fulfil this management requirement.

Multivariate analyses

Discriminant function analyses were used to develop assessment methods elsewhere. Wright, Armitage, Furse & Moss (1984) used DFA to predict reference conditions based on environmental parameters and Joy & Death (2002) used DFA to find discriminant functions of landscape variables to best differentiate among IBI classes. However, no method appears to have used DFA for metric selection (stepwise procedure) and class assignment, i.e. index computation. The advantage of this approach is that metric combinations are selected that have the highest probability of explaining biological responses to human pressures. Metrics are weighted in the discriminant function according to their individual contribution to the overall pressure–index relationship. In all IBIs developed so far, each metric receives equal weight by summing or averaging individual metric scores. However, the sensitivity of metrics may differ considerably depending on the type of metric. For example, the principle of the Multi Level Fish-based Assessment method (MuLFA; Schmutz, Kaufmann, Vogel, Jungwirth & Muhar 2000) is based on the assumption that metrics derived from higher biological organisational level (e.g. species occurrence) might react only to high doses, while metrics of lower levels (e.g. population age structure) are sensitive to lower doses of human pressures. Using stepwise procedures of multivariate statistics might solve this problem in selecting the most influential metrics. However, care is needed because there are several potential pitfalls (Quinn & Keough 2002). Stepwise procedures are dependent on defining thresholds (e.g. significance level <0.10) for selecting variables to be included in the model, but these are always arbitrary. Different stepwise techniques can produce very different final models even from the same data, impeding meaningful interpretation. In addition, the problem of collinearity among predictors in general increases with the number of variables. Therefore, uni-variate testing, pre-selection of candidate metrics and elimination of redundant metrics before applying DFA, as carried out in the proposed approach, is recommended. Automated fitting of all subsets (i.e. potential variable combinations) would be an alternative to stepwise procedures and is becoming more feasible with increasing computer power (Melcher *et al.* 2007).

Method accuracy

Accuracy of methods based on cross-validation with pre-classification varied between 47% and 98% (mean 81%) when contrasting calibration data set with degraded sites. Method accuracy did not vary among fish zones indicating consistency along the river continuum although human pressures tended to increase in an upstream–downstream continuum. Reduced availability of reference sites in lowland rivers did not hamper the development of robust methods for the barbel zone. However, only two fish types were assigned to the bream zone indicating a lack of available data in those fish types. The same method accuracy in fish types was achieved with low and high species diversity demonstrating that by selecting adequate metrics low species diversity does not undermine the development of IBIs.

Method accuracy varied considerably among ecoregions. Lowest accuracy was achieved in the ecoregion Alps. More than 90% of Alpine data were from Austria where rivers are generally only impacted by physical pressures but rarely by chemical pressures (BMLFUW 2002). Averaging physical and chemical pressures – as for the global pressure index of the pre-classification – leads to an underestimation of human pressure for Austrian rivers. Consequently, the mismatch between the pre-classification and the observed fish status is higher. Alternative approaches for defining a global pressure index for rivers impacted by a single pressure should be developed in future.

Based on a large data set, a consistent procedure for developing type-specific assessment methods could be applied to eight European ecoregions. However, splitting the database into 60 fish types impeded the method development or lowered accuracy in some of the types because of an insufficient number of remaining samples and hampered method validation with independent data. To overcome limitations of the spatially based approach at the ecoregional level merging fish types across ecoregions is proposed (Melcher *et al.* 2007).

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References

Beier U., Degerman E., Melcher A., Rogers C. & Wirlöf H. (2007) Processes of collating a European fisheries database to meet the objectives of the European Union Water Framework Directive. *Fisheries Management and Ecology* **14,** 407–416.

Belpaire C., Smolders R., Vanden Auweele I., Ercken D., Breine J., Van Thuyne G. & Ollevier F. (2000) An index of biotic integrity characterizing fish populations and the ecological quality of Flandrian water bodies. *Hydrobiologia* **434,** 17–33.

BMLFUW (2002) *Gewässerschutzbericht 2002*. Wein: Bundesministerium für Landwirtschaft, Forstwirtschaft, Umwelt und Wasserwirtschaft, 212 pp.

Breine J., Simoens I., Goethals P., Quataert P., Ercken D., Van Liefferinghe C. & Belpaire C. (2004) A fish-based index of biotic integrity for upstream brooks in Flanders (Belgium). *Hydrobiologia* **522,** 133–148.

Degerman E., Beier U., Breine J., Melcher A., Quataert P., Rogers C., Roset N. & Simoens I. (2007) Classification and assessment of degradation in European running waters. *Fisheries Management and Ecology* **14,** 417–426.

Fausch K.D., Karr J.R. & Yant P.R. (1984) Regional application of an index of biotic integrity based on stream fish communities. *Transactions of the American Fisheries Society* **113,** 39–55.

Fausch K.D., Lyons J., Karr J.R. & Angermeier P.L. (1990) Fish communities as indicators of environmental degradation. In: S.M. Adams (ed.) *Biological Indicators of Stress in Fish*, Vol. 8. Bethesda, MD: American Fisheries Society Symposium, pp. 123–144.

Ferreira T., Caiola N., Casals F., Oliveira J.M. & de Sostoa A. (2007) Assessing perturbation of river fish communities in the Iberian Ecoregion. *Fisheries Management and Ecology* **14,** 519–530.

Fritsch A.J. (1872) Die Wirbeltiere Böhmens. Ein Verzeichnis aller bisher in Böhmen beobachteten Säugetiere, Vögel, Amphibien und Fische. *Archiv für naturwissenschaftliche Landesdurchforschung von Böhmen* **2,** 1–152.

Grenouillet G., Roset N., Goffaux D., Breine J., Simoens I., de Leuuw J.J. & Kestemont P. (2007) Fish assemblages in European Western Highlands and Western Plains: a type-specific approach to assess ecological quality of running waters. *Fisheries Management and Ecology* **14,** 509–517.

Huet M. (1949) Apercu dés relations entre la pente et les populations des eaux courantes. *Schweizerische Zeitschrift fur Hydrologie* **11,** 333–351.

Hughes R.M. (1995) Defining acceptable biological status by comparing with reference conditions. In: W.S. Davies & T.P. Simon (eds) *Biological Assessment and Criteria; Tools for Water Resource Planning and Decision Making*. Boca Raton, FL: Lewis Publishers, pp. 31–47.

Hughes R.M., Howlin S. & Kaufmann P. (2004) A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. *Transactions of the American Fisheries Society* **133,** 1497–1515.

Illies J. (1967) *Limnofauna europaea. Eine Zusammenstellung aller die europäischen Binnengewässer bewohnenden mehrzelligen Tierarten mit Angaben über ihre Verbreitung und Ökologie*. Stuttgart: Gustav Fischer Verlag, 454 pp (in German.).

Joy M.K. & Death R.G. (2002) Predictive modelling of freshwater fish as a biomonitoring tool in New Zealand. *Freshwater Biology* **47,** 2261–2275.

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

de Leeuw J.J., Buijse A.D., Haidvogl G., Lapinska M., Noble R.A.A., Repecka R., Virbickas T., Wiśniewolski W. & Wolter C. (2007) Challenges in developing fish-based ecological assessment methods for large floodplain rivers. *Fisheries Management and Ecology* **14,** 483–494.

Martinez P.J., Chart T.E., Trammell M.A., Wullschleger J.G. & Bergersen E.P. (1994) Fish species composition before and after construction of a main stem reservoir on the White River, Colorado. *Environmental Biology of Fishes* **40,** 227–239.

Melcher A., Schmutz S., Haidvogl G. & Moder K. (2007) Spatially based methods to assess the ecological status of European fish assemblage types. *Fisheries Management and Ecology* **14,** 453–463.

Moss D. (2000) Evolution of statistical methods in RIVPACS. In: J.F. Wright, D.W. Sutcliffe & M.T. Furse (eds) *Assessing the Biological Quality of Fresh Waters: RIVPACS and other Techniques*. Ambleside, UK: Freshwater Biological Association, pp. 25–38.

Mundahl N.D. & Simon T.P. (1998) Development and Application of an index of biotic integrity for Coldwater Streams of the Upper Midwestern United States. In: T.S. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. Boca Raton, FL: CRC Press, pp. 383–416.

Noble R.A.A., Cowx I.G., Goffaux D. & Kestemont P. (2007) Assessing the health of European rivers using functional ecological guilds of fish communities: standardising species classification and approaches to metric selection. *Fisheries Management and Ecology* **14,** 381–392.

Noble R.A.A., Cowx I.G. & Starkie A. (2007) Development of fish-based methods for the assessment of ecological status in English and Welsh rivers. *Fisheries Management and Ecology* **14,** 495–508.

Oberdorff T. & Hughes R.M. (1992) Modification of an index of biotic integrity based on fish assemblages to characterize rivers of the Seine Basin, France. *Hydrobiologia* **228,** 117–130.

Oberdorff T., Pont D., Hugueny B. & Chessel D. (2001) A probabilistic model characterizing fish assemblages of French rivers: a framework for environmental assessment. *Freshwater Biology* **46,** 399–415.

Omernik J.M. (1987) Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* **77,** 118–125.

Omernik J.M. (1995) Ecoregions: a spatial framework for environmental management. In: W.S. Davies & T.P. Simon (ed.) *Biological Assessment and Criteria; Tools for Water Resource Planning and Decision Making*. Boca Raton, FL: Lewis Publishers, pp. 49–62.

Quinn G.P. & Keough M.J. (2002) *Experimental Design and Data Analysis for Biologists*. Cambridge: Cambridge University Press, 537 pp.

Reyjol Y., Hugueny B., Pont D., Bianco P.G., Beier U., Caiola N., Casals F., Cowx I., Economou A., Ferreira T., Haidvogl G., Noble R., De Sostoa A., Vigneron T. & Virbickas T. (2007) Patterns in species richness and endemism of European freshwater fish. *Global Ecology and Biogeography* **16,** 65–75.

Roset N., Grenouillet G., Goffaux D., Pont D. & Kestemont P. (2007) A review of existing fish assemblage indicators and methodologies. *Fisheries Management and Ecology* **14,** 393–405.

Schmutz S., Kaufmann M., Vogel B., Jungwirth M. & Muhar S. (2000) A multi-level concept for fish-based, river-type-specific assessment of ecological integrity. In: M. Jungwirth, S. Muhar & S. Schmutz (eds) *Assessing the Ecological Integrity of Running Waters*. Dordrecht: Kluwer Academic Publishers, pp. 279–289.

Smogor R.A. & Angermeier P.L. (1998) Relations between fish metrics and measures of anthropogenic disturbance in three IBI regions in Virginia. In: T.P. Simon (ed.) *Assessment Approaches for Estimating Biological Integrity using Fish Assemblages*. Boca Raton, FL: Lewis Publishers, pp. 585–610.

Smogor R.A. & Angermeier P.L. (2001) Determining a regional framework for assessing biotic integrity of Virginia streams. *Transactions of the American Fisheries Society* **130,** 18–35.

Strange R.M. (1998) Historical biogeography, ecology and fish distribution: conceptual issues for establishing IBI criteria. In: T.P. Simon (ed.) *Assessment Approaches for Estimating Biological Integrity using Fish Assemblages*. Boca Raton, FL: Lewis Publishers, pp. 65–78.

Virbickas T. & Kesminas V. (2007) Development of fish-based assessment method for the ecological status of rivers in the Baltic region. *Fisheries Management and Ecology* **14,** 531–539.

Wright J.F., Armitage P.D., Furse M.T. & Moss D. (1984) The classification of sites on British rivers using macroinvertebrates. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie* **22,** 1939–1943.

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Spatially based methods to assess the ecological status of European fish assemblage types

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Abstract A spatially based, river type-specific approach was used to develop an ecological assessment method for European rivers based on existing sampling data. The methodology comprised two main steps: (1) description of a river and fish assemblage typology based on minimally or slightly impacted sites and (2) analyses of impacted conditions for each type. Hierarchical cluster analysis of fish species assemblages identified 15 homogeneous groups in 11 European ecoregions. Discriminant analyses, based on abiotic characteristics, were used to predict fish types at impacted sites. The latter encompassed both regional (geographic position in Europe) and local factors (longitudinal zonation) influencing the distribution of riverine fishes. To assess ecological status, the responses of more than 400 metrics (species composition, abundance and age–length structure) to human pressures were tested for each river type separately. A maximum of 10 metrics per river type was selected using discriminant analysis. The density of intolerant species and feeding guilds had the highest capacity to predict the intensity of perturbation.

KEYWORDS: ecological status, Europe, metrics, modelling, rivers, Water Framework Directive.

Introduction

The spatially based approach is one option for assessing the ecological status of running waters to meet obligations under the European Union, Water Framework Directive (WFD, European Union 2000). The method classifies rivers into units with homogenous abiotic and biotic characteristics. Multimetric approaches such as the index of biotic integrity (IBI; Karr 1981) generally use an ecoregion or bioregion approach, and are thus limited to specific regions. However, Schmutz, Cowx, Haidvogl & Pont (2007a) and Schmutz, Melcher, Frangez, Haidvogl, Beier, Böhmer, Breine, Simoens, Caiola, de Sostoa, Ferreira, Oliveira, Grenouillet, Goffaux, de Leeuw, Noble, Roset & Virbickas (2007b) demonstrated that spatially based assessment methods can be developed in European ecoregions for specific fish assemblage types representing river segments with homogeneous fish assemblages. Ecoregions are supposed to provide a spatial framework for ecosystems on a large scale (Omernik & Bailey 1997) and Illies (1978) introduced a European classification system, based on 25 ecoregions. This system has not been evaluated for its ability to discriminate between fish assemblages at the continental scale, and no attempt has been made to analyse longitudinal patterns across ecoregions at the continental scale. As a result of the biogeographic situation in Europe, some assemblage types, e.g. brown trout, *Salmo trutta* L., dominated communities, can be found throughout Europe. This evidence is supported by the concept of fish zones (Huet 1949), which explains natural variability of fish communities along the longitudinal gradient of running waters.

Under the WFD, the spatially based classification of surface water bodies assumes that an abiotic river

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typology will stratify fish communities sufficiently to distinguish between natural and anthropogenic variability. However, no efforts have been made to validate this assumption at the European scale. At the ecoregional level, the spatially based approach resulted in a high number of assemblage types limiting the practical applicability of this methodology (Schmutz *et al.* 2007a, b). The potential similarity between the assemblage types identified across Europe (Reyjol, Hugueny, Pont, Bianco, Beier, Caiola, Casals, Cowx, Economou, Ferreira, Haidvogl, Noble, De Sostoa, Vigneron & Virbickas 2007) suggests that ecoregional fish types could be merged, thereby reducing the number of assessment methods necessary to monitor European rivers.

The objectives of this study were to: (1) classify homogenous fish assemblage types at the European level; (2) compare identified fish assemblage types with environmental characteristics; and (3) develop assessment methods for each identified fish assemblage type.

Methods

Data from 11 European ecoregions (12 countries) covering 1844 rivers and 5252 sites were compiled in the Fish Database of European Streams (FIDES; Beier, Degerman, Melcher & Rogers 2007) (Table 1). Data included all major river types, including the Danube, Ebro, Elbe, Garonne, Meuse, Rhine, Seine, Thames, Weser and Wisla rivers.

Fish Database of European Streams includes a number of sites with multiple fishing occasions. All sites were sampled using electric fishing during low flow periods. About 65% of sites were sampled by wading. In larger rivers (river depth >0.7 m), sampling was carried out from boats. To standardise the sampling effort, only the first run in any sampling was considered. For sites where the three run removal method was applied (2275 sites), the mean percentage of the total number of species caught during the first passage was 91.9% and the mean percentage of total abundance was 63.2% (SD

Table 1. Cross-table of abiotic WFD system A types and their occurrence within ecoregions, including the number of EFTs within each WFD type (WFD type coding: altitude (m): (1) <200, (2) 200–800, (3) >800; geological type: (1) calcareous, (2) siliceous, (3) organic; catchment size (km^2): (1) 10–100, (2) 100–1000, (3) 1000–10 000, (4) >10 000); see also *Methods*

	Ecoregion										
WFD_type (coded)	(1) Iberian Peninsula	(2) Pyrenees	(4) Alps	(8) Western Highlands	(9) Central Highlands	(13) Western Plains	(14) Central Plains	(15) Baltic Province	(18) Grat Britain	(20) Borealic Uplands	(22) Fenno-Scandian Shield
1.1.1	1			3	2	4	1	2	4		
1.1.2	1			3		5	2	2	3		
1.1.3				2	2	3	2	1	1		
1.1.4				1		2		1			
1.2.1	3			3	2	5	3	2	4		3
1.2.2	2	1		2	2	4	3	2	4		2
1.2.3	1	1		2	1	3	2	1	2		1
1.2.4				2	1	2	2				1
2.1.1	1	1	2	3	5	3			2	1	
2.1.2		2	4	4	5	4	1				
2.1.3			3	1	5	1	1				
2.1.4				2							
2.2.1	3	2	2	3	5	3	3		2	3	2
2.2.2	2	2	3	4	4	2	3			1	1
2.2.3	1		2	2	1	1	1			2	2
2.2.4				1	1						2
3.1.1		2	1		1						
3.1.2			1								
3.2.1	1	1	1	2		1				1	
3.2.2		1	1	1							
3.2.3			2								
3.2.4			1								
Number of WFD types per ecoregion	10	9	12	18	14	15	12	7	8	5	8

13.1). To avoid biases, only one randomly selected fishing occasion per site was used. The selected data set corresponds to the data used for developing the European Fish Index (EFI; Pont, Hugueny, Roset & Rogers 2007; Pont, Hugueny, Beier, Goffaux, Melcher, Noble, Rogers, Roset & Schmutz 2006).

This paper follows the principles of the spatially based methods (SBM) also described by Schmutz *et al.* (2007a, b). Predictive models were developed using biological and physico-chemical data collected at a number of non-impacted or minimally impacted sites, generally referred to as 'calibration sites'. Based on a five-tiered pre-classification of physical and chemical pressures (Degerman, Beier, Breine, Melcher, Quataert, Rogers, Roset & Simoens 2007), a set of 1455 minimally (class 1) or slightly impacted sites (class 2) was defined as the calibration data set and 3797 sites as impacted sites (classes 3, 4 and 5). The sum of the four human pressure variables, i.e. modification of morphology, hydrology, presence of toxic substances or acidification and nutrient loading, was computed. The total human pressure variable ranged from 4 to 20. This variable was re-scaled into five pressure classes; class 1: value of 4 (no pressure); class 2: values ranging from 5 to 8 (only slight pressure); class 3: 9 to 12 (moderate pressure); class 4: 13 to 16 (heavy pressure), class 5: 16 to 20 (very heavy pressure).

The principles of the SBM on European level can be summarised into four steps.

Step1: classification of calibration sites

As a prerequisite for describing reference conditions, a European fish assemblage typology was developed using calibration data only. The SBM approach at the ecoregion level revealed 60 fish assemblage types within the 11 ecoregions analysed (Schmutz *et al.* 2007a, b). These 60 fish assemblage types were clustered again using relative abundance to merge similar types across the ecoregions. Alien species were excluded in this step, whereas rarely occurring species were retained. A hierarchical cluster analysis, Ward's method (Jobson 1992), was applied using chi-squared distance as the similarity measure. The threshold for identifying distinct fish assemblages, called European fish assemblage type (EFT) was set by eye in the cluster dendrogram. The premises were: (1) to reduce the number of fish assemblages detected within individual ecoregions and (2) to have the full spectrum of fish associations along the longitudinal gradient represented.

Step 2: linking fish types with environmental variables

Seven variables were used to describe environmental characteristics of sites:

ALT	Altitude (m)
DFS	Distance from source (km)
TEM	Mean annual air temperature (°C)
LON	Longitude (WGS 84 decimal)
WID	Wetted width (m)
SLO	Slope (‰)
LAT	Latitude (WGS 84 decimal)

These variables were chosen because they describe both the regional position in the hydrographic network of European rivers and the organisation of sites along the longitudinal continuum of rivers. Autocorrelation for calibration data between the environmental variables was negligible (DFS and WID, Pearson's coefficient $r = 0.57$; TEM and ALT, $r = -0.36$; TEM and WID, $r = -0.23$; for all other combinations r was below 0.17).

All variables except TEM, LAT and LON were log-transformed [$\ln(x + 1)$]. Discriminant function analysis was used to predict EFT membership based on listed physiographic characteristics and to assign impacted sites to the respective EFT. The procedure generated a set of discriminant functions derived from combinations of the predictor variables providing the best discrimination between the groups. The functions were generated from a sample of cases for which group membership was known; they can then be applied to new cases with measurements for the predictor variables but unknown group membership. Discriminant function analysis explored the environmental distance of each test site from the centroid of each EFT in multivariate space and predicted the fish assemblage expected on the basis of a combination of environmental predictors at a test site. Critical values of Wilk's Lambda were used to determine variable entry or removal ($F = 3.84$ and 2.71) at each step of the analysis respectively.

Step 3: selecting metrics

The spectrum of the 372 tested metrics comprised overall metrics and specific guild-based metrics (Noble, Cowx, Goffaux & Kestemont 2007). Five functional metrics groups were considered: tolerance, feeding, reproduction, habitat and migration (Hughes & Oberdorff 1999; Oberdorff, Pont, Hugueny & Chessel 2001). Tolerant and intolerant groups reflect species sensitivity to any common impact related to altered habitat structure and water pollution (Noble *et al.* 2007). Loss of intolerant species is a response to degradation, whereas the number of tolerant species will tend to increase with disturbance (Pont *et al.* 2006). Additionally, 45 'sentinel species' metrics (sentinel or dominant) were tested including presence/absence density and percent of 0+ fish (Table 2).

Values for each metric per site and date were computed (Beier *et al.* 2007). Metrics were calculated systematically using both native and alien species, after having verified that metrics calculated exclusively on native species were always highly correlated ($r^2 > 0.95$) with the corresponding metric calculated for all species. All metrics were log-transformed [$\ln(x + 1)$], except relative frequencies [$\arcsin\sqrt{x}$] (Fiedler 2003). Metrics were pre-selected by eliminating metrics with insignificant pressure response using Spearman's rank correlation ($P < 0.05$).

An important task in discriminating groups is finding a set of metrics that leads to optimal differentiation and keeps the number of variables low to avoid over fitting. Stepwise discriminant function analysis was performed for each of the EFT to select metrics best discriminating between the five pressure levels. Thereby, the selection of variables was based on the significant *F*-test from an analysis of covariance. Variables already chosen, acted as covariates and the new variable was used as the dependent one. When selecting variables for entry, only one variable could be entered into the model at each step. The selection process does not take into account the relationships between variables that have not yet been selected. Thus, some important variables could be excluded in the process. Following this kind of procedure does not guarantee the selection of those variables with the lowest classification error or the lowest possible number of variables at a given error. One way to optimise metrics selection is to test all possible combinations of metrics. For this purpose, a SQL macro was developed.

Step 4: assessing new sites

A discriminant function analysis was generated for each EFT from a sample of cases for which group membership to a pressure status group was known. The minimum number of sites per class of pressure and EFT was set to 10. Pressure classes of EFT with lower numbers were excluded. After allocating new sites to

Table 2. Overview of used metrics and their abbreviations

Guild	Functional metric group	Number of species (n.sp)	Relative number of species (perc.sp)	Total density (ind/ha) (n.ha)	Relative density (%) (perc.nha)
Overall composition	All (all)	n.sp.all		n.ha.all	
Tolerance (Tol)	Intolerant (intol)	n.sp.intol	perc.sp.intol	n.ha.intol	perc.nha.intol
	Tolerant (tol)	n.sp.tol	perc.sp.tol	n.ha.tol	perc.nha.tol
Habitat (Hab)	Water Column (wc)	n.sp.Hab.wc	perc.sp.Hab.wc	n.ha.Hab.wc	perc.nha.Hab.wc
	Benthic (b)	n.sp.Hab.b	perc.sp.Hab.b	n.ha.Hab.b	perc.nha.Hab.b
	Rheophilic (rh)	n.sp.Hab.rh	perc.sp.Hab.rh	n.ha.Hab.rh	perc.nha.Hab.rh
	Limnophilic (li)	n.sp.Hab.li	perc.sp.Hab.li	n.ha.Hab.li	perc.nha.Hab.li
	Eurytopic (eury)	n.sp.Hab.eury	perc.sp.Hab.eury	n.ha.Hab.eury	perc.nha.Hab.eury
Reproduction (Re)	Lithophilic (lith)	n.sp.Re.lith	perc.sp.Re.lith	n.ha.Re.lith	perc.nha.Re.lith
	Phytophilic (phyt)	n.sp.Re.phyt	perc.sp.Re.phyt	n.ha.Re.phyt	perc.nha.Re.phyt
Longevity (Lon)	Long lived (ll)	n.sp.Lon.ll	perc.sp.Lon.ll	n.ha.Lon.ll	perc.nha.Lon.ll
	Short lived (sl)	n.sp.Lon.sl	perc.sp.Lon.sl	n.ha.Lon.sl	perc.nha.Lon.sl
Feeding (Fe)	Piscivorous (pisc)	n.sp.Fe.pisc	perc.sp.Fe.pisc	n.ha.Fe.pisc	perc.nha.Fe.pisc
	Insectivorous/Invertivorous (insev)	n.sp.Fe.insev	perc.sp.Fe.insev	n.ha.Fe.insev	perc.nha.Fe.insev
	Omnivorous (omni)	n.sp.Fe.omni	perc.sp.Fe.omni	n.ha.Fe.omni	perc.nha.Fe.omni
Migration (Mi)	Long distance (long)	n.sp.Mi.long	perc.sp.Mi.long	n.ha.Mi.long	perc.nha.Mi.long
	Potamodrom (potad)	n.sp.Mi.potad	perc.sp.Mi.potad	n.ha.Mi.potad	perc.nha.Mi.potad
45 sentinel species (e.g.)	*Esox lucius* L. (Eso.luc)			n.ha.Eso.luc	
	Salmo trutta fario L. (Sal.far)			n.ha.Sal.far	
	Cottus gobio L. (Cot.gob)			n.ha.Cot.gob	
	Leuciscus leuciscus (L.) (Leu.leu)			n.ha.Leu.leu	
	Barbatula barbatula (L.) (Bar.bab)			n.ha.Bar.bab	

Three versions were calculated, one for native, one for alien and one for all species (see Noble *et al.* 2007).

their predicted EFT on the basis of physiographic characteristics (step 2), discriminant functions (step 3) were applied to the new sites to predict their ecological status. All statistical analyses, except stepwise discriminant function analysis (SAS software), were performed using SPSS 12.0.

Validation

Models were cross-validated by comparing observed vs predicted pressure status (Joy & Death 2002). Additionally, the whole data set was randomly split into two parts, one used for model development (training data, 75%) and the other for model validation (test data, 25%) using the posterior cross-validation error rate (Fielding & Bell 1997; Hawkins, Norris, Hogue & Feminella 2000; Joy & Death 2002). The discriminant function analysis models developed with training data were applied to test data. Mean percentage of well-classified sites between training and test data were compared using a *t*-test. Normal distribution was tested with the Kolmogorov–Smirnov test.

Comparing EFT with abiotic WFD typology, system A

European Union member states are obliged to use either system A or system B of the WFD to identify distinct river types based on physiographic characteristics (WFD, Annex 1). In system A, classification variables and boundaries are pre-defined; system B is composed of compulsory and optional variables. In this study, the relation between the fish-based typology (EFT) and the pre-defined system A typology was analysed. System A typology is a hierarchical classification of running waters first assigning rivers to 25 ecoregions according to Illies (1978) and then using physiographic variables to group river segments within ecoregions. Physiographic variables, class limits and used codes were: altitude (m) [(1) <200, (2) 200–800, (3) >800]; geological type [(1) calcareous, (2) siliceous, (3) organic]; catchment size (km^2;) [(1) 10–100, (2) 100–1000, (3) 1000–10 000, (4) $>10\,000$].

Results

The full data set contained 106 fish species and 1 038 126 individuals. The calibration data set included 66 species and 373 435 individuals. The most common species occurring per sampling site was *Salmo trutta fario* L., followed by *Phoxinus phoxinus* (L.) and *Rutilus rutilus* (L.). *Rutilus rutilus* represented the most abundant species in the whole calibration data set (13.8%) followed by *Gobio gobio* (L.), *Barbatula barbatula* (L.), *P. phoxinus* and *S. trutta fario* (8.3%). On average, 5.3 fish species occurred per sampling site.

European fish assemblage types (EFT)

The 60 fish assemblage types identified at the ecoregional level were merged into 15 EFT (Fig. 1). The dendrogram first split the data into two clusters; cluster A (types 1–3) represents headwater assemblages with low species richness and cluster B is characterised by more diverse fish communities (see Table 3 for their relative native species composition). Types 1 to 4 are streams dominated by *S. trutta fario* and varying in the amount of accompanying species. Types 5 and 6 represent downstream river sections with lower gradient dominated by *P. phoxinus* (Fig. 2). Type 5 differs from type 6 mainly by its higher proportion of *Anquilla anquilla* (L.). *Thymallus thymallus* (L.) is mainly found in types 7 and 9, but with the latter from northern Europe. Types 8, 11 and 12 are dominated by anadromous or potamodromous salmonids, i.e. *Salmo salar* L., *Salmo trutta lacustris* L., *Salmo trutta trutta* L. Types 10 and 13 represent southern fish assemblages, with the latter characterised by Mediterranean endemics. Types 14 and 15 are lowland rivers dominated by *Gasterosteus aculeatus* L. and *R. rutilus*. Types 7, 9, 10, 11 and 14 were represented only in one ecoregion. Others like types 6 or 15 are dispersed over five ecoregions. Types 9 and 11 (n = 7 and 9) were excluded from further analyses because of low number of sites.

Linking EFT with abiotic variables

All seven environmental variables (i.e. ALT, SLO, TEM, WID, DFS, LAT and LON) were selected by the analysis. The eigenvalues calculated by the discriminant function analysis reflect the model variance

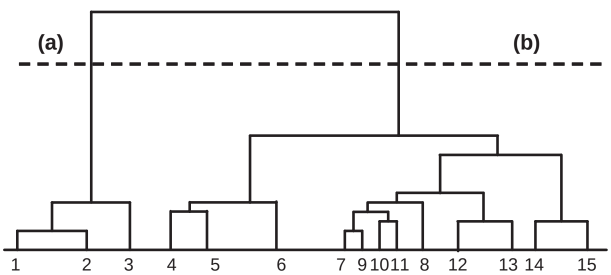

Figure 1. Dendrogram (simplified) depicting cluster analyses of 15 European fish types (EFTs) using relative percentage of occurrence. Horizontal dashed line shows arbitrary division line for defining clusters A and B.

Table 3. The 15 European fish types (EFT), mean number of native fish species and mean relative species composition (%); only species occurring >2% are listed (species >12% are in bold)

Fish species	European fish types (EFT)														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Number of sites	157	365	553	229	1130	832	69	84	7	81	9	446	148	67	432
Mean number of species per sample	1	2	4	4	5	8	4	4	3	4	1	3	2	12	9
Salmo trutta fario L.	**94**	**81**	**43**	**37**	11	5	**45**	7	**25**	14		9	4		3
Cottus gobio L.		**14**	**38**	5	**19**	**12**	4	**13**				**17**			
***Phoxinus phoxinus* (L.)**			7	**17**	**21**	**31**		9				7	**15**	3	2
Barbatula barbatula (L.)			3	**13**	**14**	**13**									3
Anguilla anguilla (L.)	3				**16**					3			9		
Leuciscus souffia Risso				12											
Thymallus thymallus (L.)							**45**	11	**18**						
***Salmo salar* L.**	2				7			**45**	9			3			
***Cottus poecilopus* Heckel**		2						5	**47**						4
***Leuciscus carolitertii* Doadrio**										**36**					
Chondrostoma polylepis Steindachner										**23**					
Rutilus arcasii (Steindachner)										**14**					
Barbus bocagei Steindachner										10					
Salmo trutta lacustris L.											**100**	6			2
Salmo trutta trutta L.												**40**			
***Barbus meridionalis* Risso**													**53**		
Leuciscus cephalus (L.)				4	2	5		2					11	10	8
Barbus haasi Mertens													8		
***Gasterosteus aculeatus* L.**					2								0	**39**	
Alburnoides bipunctatus (Bloch)						3								**15**	
***Rutilus rutilus* (L.)**					3	6						2		10	**37**
Alburnus alburnus (L.)						4								6	7
Gobio gobio (L.)				6		5								4	7
Perca fluviatilis L.						3						4			6
Lota lota (L.)												4			2
Leuciscus leuciscus (L.)						4								2	4
Esox lucius L.															3
Barbus barbus (L.)															2

explained. The relative contribution of each function to the overall model was 49.9% for the first function, 21.4% for the second, 15.6% for the third and 9.3% for the fourth accounting for 96.1% of the variance (Table 4). The largest absolute correlation between discriminant functions and abiotic variables existed for ALT and SLO (function 1), LON and TEM (function 2) and LAT and DFS (function 3, Table 5).

Cross-validation, ratio between observed and expected type, revealed that 69.9% of the sites were correctly assigned to their predetermined fish type (Fig. 3). The 75% quartile ranged from 60% to 100%.

Comparing EFT and WFD typology, system A

Applying system A of the WFD to the data set resulted in 118 different types (Table 5). The minimum number of abiotic types within a single ecoregion was 5 and the maximum 18 types. This is compared with 15 EFTs, ranging from one to five EFTs within single WFD types.

Selecting metrics and assessing the ecological status

In total, 115 metrics from 372 potential metrics showed significant ($P < 0.05$, Spearman's rank correlation) responses to pressure. Finally, 44 different metrics were selected in the 13 discriminant function analysis models. The number of metrics per EFT varied from 3 to 10. 'Density of intolerant species' was the most common metric across all models and was used in seven of 13 EFTs. Four other metrics, i.e. 'density of omnivorous species', 'number of insectivorous species', 'number of omnivorous species' and 'percentage of long distance migratory species' were used in four EFTs. About 43% of the metrics were specific to individual EFTs.

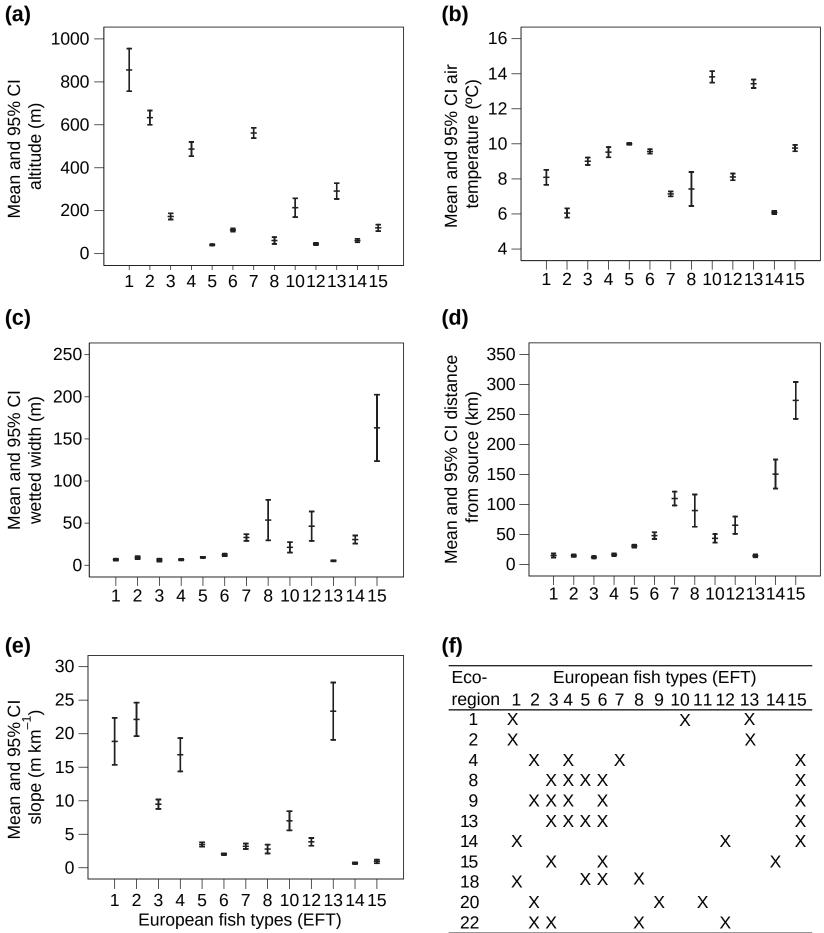

Figure 2. Mean and 95% confidence intervals (CI) for abiotic descriptors of the European fish types: (a) altitude; (b) annual air temperature; (c) wetted width; (d), distance from source; (e) slope and (f) assignment of European fish types to ecoregions.

Table 4. Eigenvalues, variance and canonical correlation of discriminant function analysis used for EFTs

Function	Eigenvalue	% of variance	Cumulative %	Canonical correlation
1	1.96	49.87	49.87	0.81
2	0.84	21.39	71.26	0.68
3	0.61	15.58	86.84	0.62
4	0.37	9.30	96.14	0.52
5	0.09	2.27	98.42	0.29
6	0.06	1.43	99.84	0.23
7	0.01	0.16	100.00	0.08

Table 5. Factor structure matrix of the seven significant discriminant functions used to predict EFT membership based on environmental variables (largest absolute correlation between each variable and any discriminant function in bold)

	Function						
Descriptor	1	2	3	4	5	6	7
ALT	**−0.716**	0.186	−0.003	−0.198	0.494	0.386	−0.146
SLO	**−0.675**	−0.196	−0.405	0.280	−0.222	0.251	0.389
LON	−0.097	**0.871**	−0.179	0.405	0.151	−0.112	−0.031
TEM	0.119	**−0.686**	0.647	0.237	−0.074	−0.136	−0.125
LAT	0.418	0.404	**−0.687**	0.102	0.049	0.348	0.238
DFS	0.256	0.371	**0.571**	−0.284	−0.202	0.156	0.570
WID	0.190	0.375	0.451	−0.278	**−0.620**	0.395	0.050

The most frequently selected functional groups were insectivorous and omnivorous species (used nine times) followed by intolerant species (eight times) (Table 6). Feeding and habitat guilds were used most often. Metrics responded in most cases in one direction only, i.e. consistently either increased or decreased in all EFTs. However, metrics referring to migration, longevity, piscivorous species and sentinel species (*S. trutta fario*) both increased and decreased with disturbance (Table 6).

The proportion of correctly classified sites (cross-validation of observed against predicted), when

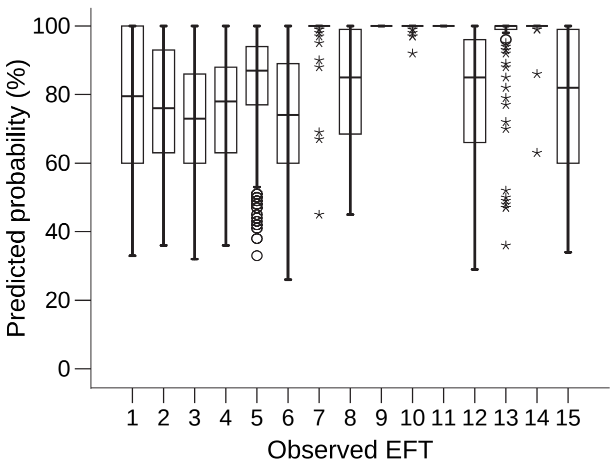

Figure 3. Box-plots of predicted probability of being a specific EFT (circles – outliers, stars – extreme values).

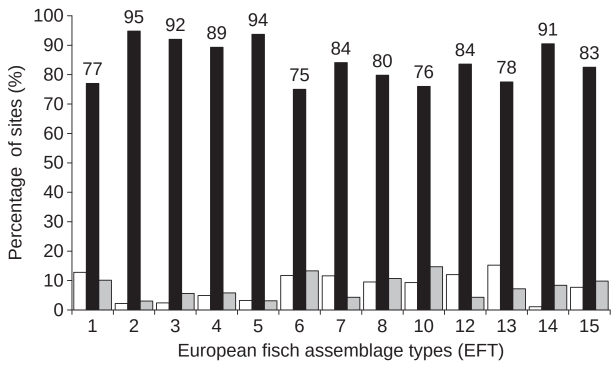

Figure 4. Percentage of well-classified and mis-classified sites according to the predicted ecological status (impacted or unimpacted). ☐ unimpacted sites misclassified as impacted, ■ correctly classified, ■ impacted sites misclassified as unimpacted.

Table 6. Frequency of metrics used per guild category and functional metric group

Guild	Frequency per guild category	Functional metric group	Frequency per metric group	Response
Feeding	21	insev	9	−1
		omni	9	1
		pisc	3	1 (2), −1 (14, 15)
Habitat	16	b	2	−1
		eury	2	1
		li	3	1
		rh	4	−1
		wc	5	1
Migration	13	long	7	1 (4, 15), −1 (1, 5, 8, 14)
		potad	6	1 (2, 8), −1 (3, 7, 15)
Tolerance	10	intol	8	−1
		tol	2	1
Sentinel species	10	Eso.luc	3	1
		Sal.far	2	1 (8), −1(5)
		Cot.gob	2	−1
		Leu.leu	2	1
		Bar.bab	1	1
Longevity	8	ll	4	1 (1, 3), −1 (10, 13)
		sl	4	1 (4, 5), −1 (7, 15)
Reproduction	7	phyt	5	1
		lith	2	−1
Overall composition	2	all	2	−1

Response (−1 = negative, 1 = positive; in brackets EFT number) of guilds and functional metric groups used for the assessment methods (For metric abbreviations see Table 2).

comparing impacted (pressure class > 2) vs unimpacted sites (pressure class 1 or 2), was on average 84.5% (75–95%) of all sites (training and test data) (Fig 4). A two-class system was used because there were not data for all five pressure classes and for each EFT available. For independent validation, the comparison between cross-validations showed no significant difference ($t = -0.517$, $P = 0.610$) between the level of correctly classified sites in the training (83.8%) and test data set (85.3%).

Differences between observed and predicted pressure status varied among 12 countries (chi-squared test, $P < 0.001$). The percentage of well-predicted sites ranged from 97.8 (Belgium-Flanders), 94.8 (the Netherlands), 90.8 (Belgium-Wallonia), 90.5 (Sweden), 85.8 (Austria), 83.5 (Austria), 83.5 (France), 81.4 (UK), 80.9 (Lithuania), 76.1 (Portugal), 75.0 (Spain), 71.9 (Poland) to 67.8 (Germany).

Discussion

This study developed a fish-based river typology and fish-based assessment methods for the ecological status of fish assemblage types at the European scale. The typology developed for each ecoregion (Schmutz *et al.* 2007a, b) was used to determine the fish assemblages at the continental scale. The number of types was reduced from 60 ecoregional types to 15 EFTs. A model was also developed that uses environmental characteristics to predict fish assemblage types for new sites. Most of the variation in physiographical variables of EFT was accounted for by upstream–downstream gradients (width, slope and distance from source and altitude). These gradients reflect an increase in fish species richness with increasing catchment size. Headwaters appeared to be similar across Europe and dominated by salmonid species regardless of geographical position in Europe. The large-scale geographical pattern in fish assemblage shows that most of the EFTs are spread across Europe, only four EFTs were restricted to single ecoregions, e.g. EFTs with Mediterranean endemics occur only in southern Europe. Only two lowland fish assemblages (EFT 14 and 15) were identified, indicating a lack of reference sites of lowland rivers in the data set. Fish assemblage types dominated by large rheophilic cyprinids (*Chondrostoma* sp., *Barbus* sp.) forming the classical barbel fish zone (Thienemann 1925) are missing in the database. Lowland rivers are under high pressure throughout Europe and hence reference sites are very scarce. The same is true for very large rivers (Fig. 2). In addition, the east–west gradient in species richness across Europe (Banarescu 1989; Reyjol *et al.* 2007) is not reflected in the EFTs. More data from lowland and large rivers, in particular from Eastern ecoregions, are necessary to fill these gaps.

Seven environmental descriptors were selected by discriminant function analysis to predict EFTs. Function 1 mainly described longitudinal aspects (ALT, SLO), function 2 focused on regional descriptors (TEM, LON), functions 3 and 5 combined both spatial dimensions (DFS, LAT and WID). Joy & Death (2002) tested various environmental variables to discriminate among fish assemblages and found that altitude and distance from sea were the most important factors associated with fish community structure, while local scale variables were less important. Also other studies (e.g. Jowett & Richardson 1996) showed that catchment-scale variables were the most important driving fish assemblage structures. Pont, Hugueny & Oberdorff (2005) found three environmental descriptors at the local scale (river slope, river width and upstream drainage area) and three at regional scale (mean annual air temperature, mean annual air temperature range and basin unit) predicted the occurrence of fish species.

In this study, about 70% of the sites were assigned correctly to their pre-determined fish types; this level of precision was similarly reported in the literature for fish (Joy & Death 2002) and macroinvertebrates (Turak, Flack, Norris, Simpson & Waddell 1999).

A total of 118 WFD types were classified using the WFD system A and FIDES data set, of which 36% were similar between WFD types and EFTs. However, in 64% of cases, there were up to five EFTs within individual WFD types. This shows that although the WFD system A employs many more types it does not differentiate fish assemblages adequately along the longitudinal gradient.

Fish-based assessment method

The concept of spatially based assessment methods is to test metrics response for each identified EFT (Schmutz, Kaufmann, Vogel, Jungwirth & Muhar 2000). This study showed that the number and type of 44 selected metrics differed among EFTs. The number of metrics ranged from three to 10 metrics. For SBM at ecoregion level, 130 metrics were used and per type a median number of 9.2 metrics was selected (Schmutz *et al.* 2007a, b). At the European scale, the EFI employs 10 metrics (Pont *et al.* 2007) compared with seven metrics in a similar index developed for France (Oberdorff, Pont, Hugueny & Porcher 2002).

The proposed standardised procedure provided the opportunity to examine the relative importance of two metric types, i.e. functional guilds (e.g. habitat and reproduction) and sentinel species (e.g. 0 +). All but six models (EFT 2, 3, 5, 6, 8, 12) incorporated both functional guilds and sentinel species. Only five sentinel species were selected. *Cottus gobio* L., a species supposedly very sensitive to human perturbation (Alabaster & Lloyd 1982), decreased with human pressure. *Salmo trutta fario* increased or decreased depending on the EFT. *Esox lucius* L., *Leuciscus leuciscus* (L.) and *B. barbatula* increased with human pressures. This shows that the reaction of metrics to human pressures may vary and depends on the specific EFTs it was selected for.

The abundance of intolerant species was the most important metric. Intolerance was defined as sensitivity to eutrophication, acidification and habitat degradation (Noble *et al.* 2007). This also indicates that water pollution is still a major problem in European running waters and that fish are sensitive to this kind of human pressure. Other frequently selected metrics refer to feeding and habitat requirements, while metrics describing overall composition had very poor discriminative capacity.

Metrics referring to migration (long migration and potamodromous) and longevity (long living and short living) showed no clear pattern when compared among EFTs, i.e. increased or decreased with human pressures. The inconsistent reaction of migratory species may be because the human pressure index used did not include dams and weirs. Short-lived species such as *P. phoxinus* or *B. barbatula* might indicate human pressure by either a decrease in EFTs where they are dominant (EFT 4, 5 and 6) or an increase in EFTs where they are less abundant but replace dominant species.

A source of bias is the pressure pre-classification because it was partly based on expert judgement, in particular the quantification of physical pressures (morphological and hydrological). The accuracy of these classifications was probably not always sufficient and not fully standardised among countries (Degerman *et al.* 2007). To define a general human pressure index, the values of four human pressure variables were summed and rescored into five discrete pressure classes. The level of resolution is hence defined by the five classes, i.e. there are no between-class or finer gradients in the pressure classification.

In developing fish indices, the responses to the pre-classification of human pressures were optimised using discriminant function analyses. The problem of over fitting the models was overcome by limiting the number of metrics selected per model. An independent validation showed no difference between the training and test data sets. The models were consistent under varying environmental conditions, i.e. there was no difference in misclassification between headwater and lowland assemblages or salmonid and cyprinid rivers but there were differences among different countries.

In conclusion, a fish-based river typology and standardised indices of biotic integrity at the Europe scale were developed in accordance with the WFD. It is worth noting that besides the EFI (Pont *et al.* 2006), this study calibrated fish indices by a pre-classification approach at European scale. This tool provides the basis for a standardised ecological assessment of European rivers.

One main output, the EFT classification, is part of the EFI software that can be downloaded from http://fame.boku.ac.at/. The assessment method developed helped to validate the EFI with other methods (Schmutz *et al.* 2007a, b). At present, the approach applies only to rivers belonging to the 13 EFTs defined in the present work. To extend the geographic range of the method and to include new fish assemblage types, additional data have to be collected and the analyses repeated.

Acknowledgements

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References

Alabaster J.S. & Lloyd R. (1982) *Water Quality Criteria for Freshwater Fish*, 2nd edn. Butterworths, London: Food and Agriculture Organisation of the United Nations, 361 pp.

Banarescu P. (1989) Zoogeography and history of the freshwater fish fauna of Europe. The freshwater fishes of Europe. In: J. Holčik (ed.) *General Introduction to Fishes, Acipenseriformes*, Vol. 1. Wiesbaden: AULA-Verlag, pp. 88–107.

Beier U., Degerman E., Melcher A., Rogers C. & Wirlöf H. (2007) Processes of collating a European fisheries database to meet the objectives of the European Union Water Framework Directive. *Fisheries Management and Ecology* **14,** 407–416.

Degerman E., Beier U., Breine J., Melcher A., Quataert P., Rogers C., Roset N. & Simoens I. (2007) Classification and assessment of degradation in European running waters. *Fisheries Management and Ecology* **14,** 417–426.

EU Water Framework Directive (2000) Directive 2000/60/EC of the European Parliament and the Council of 23 October 2000 establishing a framework for community action in the field of water policy. *Official Journal of the European Communities* (22.12.2000) **L327**, 1.

Fiedler K. (2003) *Elementare statistische Methoden für Biologen*. Vorlesungsunterlagen: Universität Wien, 53 pp.

Fielding A.H. & Bell J.F. (1997) A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation* **24,** 38–49.

Hawkins C.P., Norris R.H., Hogue J.N. & Feminella J.W. (2000) Development and evaluation of predictive models for measuring the biological integrity of streams. *Ecological Applications* **10,** 1456–1477.

Huet M. (1949) Apercu dés relations entre la pente et les populations des eaux courantes. *Schweizerische Zeitschrift fur Hydrologie* **11,** 333–351.

Hughes R.M. & Oberdorff T. (1999) Applications of IBI concepts and metrics to waters outside the United States and Canada. In: T.P. Simon (ed.) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. Boca Raton, FL: Lewis Press, pp. 79–83.

Illies J. (1978) *Limnofauna Europaea. A Check-list of the Animal Inhabiting European Inland Waters, with an Account of their Distribution and Ecology*, 2nd edn. Stuttgart: Gustav Fischer Verlag, 241 pp.

Jobson J.D. (1992) *Applied Multivariate Data Analysis. Volume II: Categorical and Multivariate Methods*. New York: Springer Verlag, 731 pp.

Jowett I.G. & Richardson J. (1996) Distribution and abundance of freshwater fish in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* **30,** 239–255.

Joy M.K. & Death R.G. (2002) Predictive modelling of freshwater fish as a biomonitoring tool in New Zealand. *Freshwater Biology* **47,** 2261–2275.

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

Noble R.A.A., Cowx I.G., Goffaux D. & Kestemont P. (2007) Assessing the health of European rivers using functional ecological guilds of fish communities: standardising species classification and approaches to metric selection. *Fisheries Management and Ecology* **14,** 381–392.

Oberdorff T., Pont D., Hugueny B. & Chessel D. (2001) A probabilistic model characterizing fish assemblages of French rivers: a framework for environmental assessment. *Freshwater Biology* **46,** 399–415.

Oberdorff T., Pont D., Hugueny B. & Porcher J.P. (2002) Development and validation of a fish-based index for the assessment of 'river health' in France. *Freshwater Biology* **47,** 1720–1734.

Omernik J.M. & Bailey R.G. (1997) Distinguishing between watersheds and ecoregions. *Journal of the American Water Resources Association* **33,** 935–949.

Pont D., Hugueny B. & Oberdorff T. (2005) Modelling habitat requirement of European fishes: do species have similar responses to local and regional environmental constraints? *Canadian Journal of Fisheries and Aquatic Sciences* **62,** 163–173.

Pont D., Hugueny B., Beier U., Goffaux D., Melcher A., Noble R., Rogers C., Roset N. & Schmutz S. (2006) Assessing river biotic condition at a continental scale: a European approach using functional metrics and fish assemblages. *Journal of Applied Ecology* **43,** 70–80.

Pont D., Hugueny B. & Rogers C. (2007) Development of a fish-based index for the assessment of river health in Europe: the European Fish Index. *Fisheries Management and Ecology* **14,** 427–439.

Reyjol Y., Hugueny B., Pont D., Bianco P. G., Beier U., Caiola N., Casals F., Cowx I., Economou A., Ferreira T., Haidvogl G., Noble R., de Sostoa A., Vigneron T. & Virbickas T. (2007) Patterns in species richness and endemism of European freshwater fish. *Global Ecology and Biogeography* **16,** 65–75.

Schmutz S., Kaufmann M., Vogel B., Jungwirth M. & Muhar S. (2000) A multi-level concept for fish-based, river-type-specific assessment of ecological integrity. *Hydrobiologia* **422/423,** 279–289.

Schmutz S., Cowx I.G., Haidvogl G. & Pont D. (2007a) Fish-based methods for assessing European running waters: a synthesis. *Fisheries Management and Ecology* **14,** 369–380.

Schmutz S., Melcher A., Frangez C., Haidvogl G., Beier U., Böhmer J., Breine J., Simoens I., Caiola N., de Sostoa A., Ferreira M.T., Oliveira J., Grenouillet G., Goffaux D., de Leeuw J.J., Noble R.A.A., Roset N. & Virbickas T. (2007b) Spatially based methods to assess the ecological status of riverine fish assemblages in European ecoregions. *Fisheries Management and Ecology* **14,** 441–452.

Thienemann A. (ed.) (1925) *Die Binnengewässer Mitteleuropas. Eine limnologische Einführung. Die Binnengewässer.* Stuttgart: Schweitzerbart'sche Verlagsbuchhandlung.

Turak E., Flack L. K., Norris R. H., Simpson J. & Waddell N. (1999) Assessment of river condition at a large spatial scale using predictive models. *Freshwater Biology* **41,** 283–298.

Appendix

Annex 1: Final list of selected 44 fish metrics for each European fish assemblage type (EFT) used for the assessment methods, listed due to their frequency of occurrence (metric abbreviation see Table 1).

	EFT												
Metrics selected	1	2	3	4	5	6	7	8	10	12	13	14	15
n.ha.intol		×						×	×	×	×	×	×
n.sp.Fe.insev						×	×		×	×			
perc.sp.Mi.long	×			×	×			×					
n.ha.Fe.omni		×		×	×					×			
n.sp.Fe.omni		×	×									×	×
n.ha.Eso.luc					×	×				×			
n.ha.Fe.insev	×		×	×									
perc.sp.Lon.ll			×						×		×		
n.ha.Re.phyt		×						×			×		
n.ha. Fe.pisc		×										×	×
n.ha.Mi.potad							×	×		×			
n.ha.Hab.b												×	×
n.ha.Cot.gob								×		×			
perc.sp.Fe.insev				×								×	
n.ha.Leu.leu			×			×							
n.sp.Hab.li					×	×							
n.sp.Re.lith			×		×								
n.ha.Mi.long												×	×
n.sp.Mi.potad			×										×
n.sp.Hab.rh												×	×
n.ha.Sal.far					×			×					
perc.sp.Lon.sl				×			×						
n.sp.Hab.wc	×									×			
perc.sp.Hab.wc							×	×					
density.sp.all						×							
n.sp.all						×							
n.ha.Bar.bar		×											
n.sp.Hab.eury										×			
perc.sp.Hab.eury					×								
n.sp.intol												×	
perc.sp.Hab.li							×						
n.sp.Lon.ll	×												
n.sp.Mi.long								×					
perc.sp.Fe.omni					×								
n.sp.Re.phyt											×		
perc.sp.Re.phyt							×						
perc.nha.Mi.potad		×											
n.ha.Hab.rh										×			
perc.sp.Hab.rh											×		
n.ha.Lon.sl													×
n.sp.Lon.sl					×								
n.ha.tol				×									
perc.sp.tol						×							
n.ha.Hab.wc					×								

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Assessing river biotic condition at a continental scale: a European approach using functional metrics and fish assemblages

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Summary

1. The need for sensitive biological measures of aquatic ecosystem integrity applicable at large spatial scales has been highlighted by the implementation of the European Water Framework Directive. Using fish communities as indicators of habitat quality in rivers, we developed a multi-metric index to test our capacity to (i) correctly model a variety of metrics based on assemblage structure and functions, and (ii) discriminate between the effects of natural vs. human-induced environmental variability at a continental scale.

2. Information was collected for 5252 sites distributed among 1843 European rivers. Data included variables on fish assemblage structure, local environmental variables, sampling strategy and a river basin classification based on native fish fauna similarities accounting for regional effects on local assemblage structure. Fifty-eight metrics reflecting different aspects of fish assemblage structure and function were selected from the available literature and tested for their potential to indicate habitat degradation.

3. To quantify possible deviation from a 'reference condition' for any given site, we first established and validated statistical models describing metric responses to natural environmental variability in the absence of any significant human disturbance. We considered that the residual distributions of these models described the response range of each metric, whatever the natural environmental variability. After testing the sensitivity of these residuals to a gradient of human disturbance, we finally selected 10 metrics that were combined to obtain a European fish assemblage index. We demonstrated that (i) when considering only minimally disturbed sites the index remains invariant, regardless of environmental variability, and (ii) the index shows a significant negative linear response to a gradient of human disturbance.

4. *Synthesis and applications*. In this reference condition modelling approach, by including a more complete description of environmental variability at both local and regional scales it was possible to develop a novel fish biotic index transferable between catchments at the European scale. The use of functional metrics based on biological attributes of species instead of metrics based on species themselves reduced the index sensitivity to the variability of fish fauna across different biogeographical areas.

Key-words: biotic river integrity, community structure, European fish fauna, habitat disturbance, modelling, Water Framework Directive

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Introduction

Fish populations and communities are sensitive indicators of habitat quality in rivers because they react significantly to almost all kinds of anthropogenic disturbances, including eutrophication, acidification, chemical pollution, flow regulation, physical habitat alteration and fragmentation (reviewed by Ormerod 2003). This sensitivity to the relative health of their aquatic environments and the surrounding watersheds is the basis for using biological monitoring of fishes to assess environmental degradation (Fausch *et al*. 1990). Over the last 30 years, a variety of fish-based biotic indices have been widely used to assess river quality, and the use of biologically based multimetric indices, inspired by the index of biotic integrity (IBI) (Karr 1981; Karr *et al*. 1986), has grown rapidly (Simon 1999). The main characteristic of these tools is that they employ a series of metrics based on assemblage structure and function that are integrated into a numerical index scaled to reflect the ecological health of the assemblage. Another characteristic is that they use the 'reference condition approach' (Bailey *et al*. 1998), comparing an ecosystem exposed to a potential stress with a reference system unexposed to such a stress (Hughes *et al*. 1998).

The accuracy of these biological assessments depends primarily on the sensitivity of these tools to natural environmental variation as opposed to human-induced disturbances of river biota. To reduce or remove the confounding effects of natural environmental variability, most authors have validated indices over a restricted range of geographical and environmental situations: particular states (Roth *et al*. 1998; Schleiger 2000), ecoregions and drainage areas (McCormick *et al*. 2001; Smogor & Angermeier 2001; Emery *et al*. 2003; Mebane, Maret & Hughes 2003), river sizes (Angermeier & Schlosser 1987; Simon & Emery 1995), water thermal regimes (Leonard & Orth 1986; Hughes, Howlin & Kaufmann 2004) and levels of fish diversity (Harris & Silveira 1999). Most authors also account for between-site natural variability by standardizing metrics in relation to river size. Angermeier, Smogor & Stauffer (2000) consider that multimetric indices perform best when coupled with a regional framework so that the metrics reflect region-specific attributes of natural biotic communities. Hughes, Whittier & Larsen (1990) called for a necessary compromise between the extremes of uniform nation-wide criteria and unique criteria for each waterbody. However, regardless of the approach, our capacity to distinguish between natural and human-induced variation of biological conditions at both local and regional scales remains a crucial point (Hughes *et al*. 1998).

The new European Union (EU) water policy, the Water Framework Directive (WFD), states that all European rivers should be assessed via a reference condition approach using bioassessment tools based on four biotic elements, including fish (EU 2000). These biological assessment tools must also indicate which functional characteristics of the biota are altered, in order to increase the probability of success of ecological river rehabilitation schemes (Pretty *et al*. 2003; Giller 2005; Palmer *et al*. 2005).

One way to attain this goal is to develop a common assessment method at the European scale using defined metrics that remain insensitive to natural environmental variability for all unimpaired sites, and that are monotonically linked to the intensity of human alteration for impaired sites. The objective of this present study was to develop a fish-based index applicable to all European rivers using a methodology already tested at a national level in France (Oberdorff *et al*. 2001, 2002). We had two main questions. (i) Is it possible, at the European scale, to model correctly a variety of metrics as a function of natural environmental descriptors defined at both local and regional scales, in the absence of any human disturbance? (ii) Are we able to quantify, for any tested site, its deviation from a reference condition site having similar natural environmental conditions?

Methods

SITE SELECTION AND PRE-CLASSIFICATION OF DISTURBANCES

We used data from fish surveys of 12 European countries conducted by several laboratories and governmental environmental agencies (1978–2002). These 5252 river reaches or sites (Fig. 1) cover most of the climatic and physical conditions that occur in Europe.

All sites had been sampled using electrofishing techniques (DC or PDC (pulsed direct current) waveform) during low flow periods. When possible (river depth $< 0{\cdot}7$ m), river reaches were sampled by wading (64·9% of all sites). For most of these sites, the removal method was applied (87·9%) and stops nets were not used (88·7%). In large rivers (river depth $> 0{\cdot}7$ m) sampling was from boats (35·1% of all sites), mainly in near-shore areas. The size of each sampled site was sufficient to encompass complete sets of characteristic local river habitat. For 67·7% of all sites, the whole river width was sampled. In others cases, the whole river section was only partially sampled (mainly in near-shore areas). In order to standardize the sampling effort, we only considered the first passage in all cases. Although our data are subjected to sampling noise, this sampling effort was sufficient to describe the fish assemblage. For sites where the removal method was applied with three successive passages (2275 sites), the mean percentage of the total number of species caught during the first passage was 91·9% (SD 16·3%) and the mean percentage of total abundance was 63·2% (SD 13·1%). Sampling effort was summarized by three variables: sampling technique (TECH; boat or wading), sampling method (METH; complete, whole river width sampling, or partial) and fished area (FISH). We only retained one fishing occasion per site.

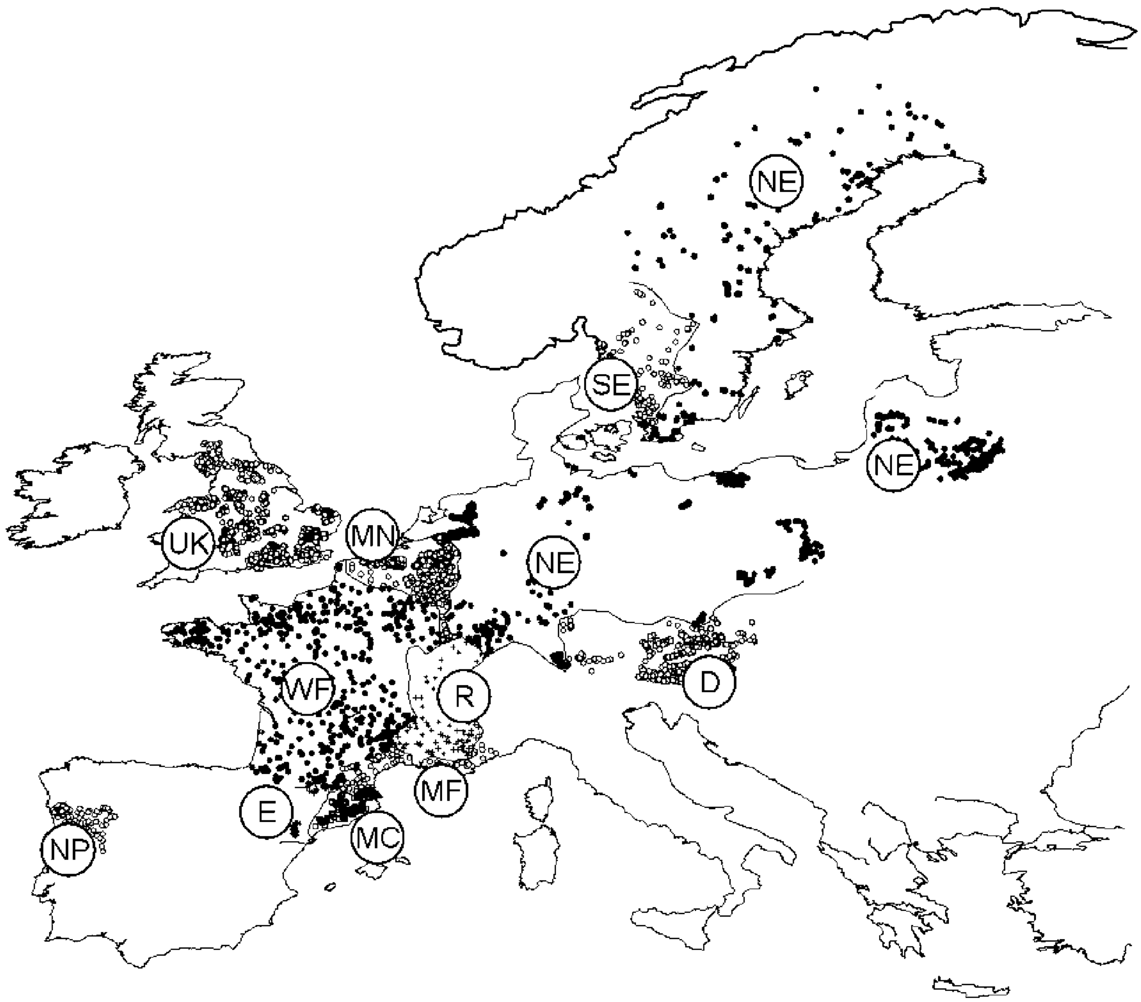

Fig. 1. Map of Europe showing 11 river groups and the 5252 sites. D, Danube; E, Ebro River; MC, Mediterranean rivers from Catalunya; MF, Mediterranean rivers from France; MN, Meuse-group rivers; NP, north Portugal rivers; NE, northern European plain rivers; R, Rhône River; SE, south-west Sweden rivers; UK, United Kingdom rivers; WF, west France rivers. Symbols (circles and pluses) are only used to distinguish between river groups.

For each site, the degree of human-induced alterations was evaluated based on available data, existing knowledge and expert judgement. Four disturbance variables were retained and rated as a function of their deviation from a natural state (from 1, no deviation, to 5, heavily degraded): hydrological disturbances (HYDR; classes 1–5, from more than 90% to less than 50% of the mean natural water level and from almost no to strong deviation from natural duration and intensity of flooding period); morphological conditions (MORPH; from negligible morphological alteration to complete channelization with most natural habitats missing); phosphorous, nitrogen and total organic carbon (NUTR; from conditions within 150% of background levels to deviation more than 300% from national backgrounds levels); and deviation from critical values of, for example, oxygen and pH (TOX). A total disturbance assessment (DISTURB) was obtained by summing up the four disturbance variables (range from 4 to 20).

Among the 5252 sites, 1608 sites were considered as reference sites (REF) when none of the four disturbance variables were rated over 2. This definition ensured sufficient sample sizes for all countries. While not all reference sites were pristine or totally undisturbed, the degree of alteration was null or very low. Among others sites, we distinguished between weakly disturbed (WI sites; 8 < DISTURB < 13) and heavily disturbed sites (HI sites; DISTURB > 12).

Each metric-specific model and the final index were validated independently by randomly dividing the reference data set into three subsets used for model calibration (REF-CAL, 1000 sites), model validation (REF-MET, 304 sites) and final index validation (REF-IND). We also randomly selected among the weakly disturbed sites two sets for metric selection (WI-MET, 958 sites) and final index validation (WI-IND, 304 sites) and among the heavily disturbed sites one set for metric selection (HI-MET, 958 sites) and one set for final index validation (HI-IND, 304 sites).

LOCAL ENVIRONMENTAL VARIABLES

Nine abiotic variables were measured in the field or from topographical maps, or estimated using GIS at each site: altitude (ELE; 0–1950 m), distance from source (DIS; 0–990 km), basin class (CAT; < 10 km^2, 10–99 km^2, 100–999 km^2, 1000–9999 km^2, > 10 000 km^2), reach slope (SLOP; 0·01–199 m km^{-1}), wetted width (WID; 0·5–1600 m), mean annual air temperature (TEMP; −2–+16 °C), presence/absence of a natural lake upstream (LAK), geological type (GEO; calcareous, siliceous) and flow regime (FLOW; permanent or temporary). Four of these explanatory variables (ELE, SLOP, DIS, WID) and FISH were log-transformed to reduce the skewness of their distribution.

REGIONAL UNITS

To delineate biologically relevant regional units for European fish, we first considered the complete fish fauna lists for each drainage basin unit, which represent homogeneous entities with regard to long-term dispersal (Matthews 1998) and explain a significant part of fish community variability (Pont, Hugueny & Oberdorff 2005). However, because of the lack of available data for small basins, we grouped all coastal basins smaller than 25 000 km^2 and draining to a given sea coast (ICES Fishing Areas, http://www.ices.dk), hypothesizing that these contiguous basins were in contact during recent Holocene sea level variations. For our entire study area, we then compiled data from previous literature to establish lists of native fish species of 19 large basins and 17 groups of contiguous small basins. We examined the similarities (Jaccard Index) between these 36 fauna lists using the unweighted arithmetic average clustering method. As a cut-off value, we chose the similarity level corresponding to the best compromise between a minimal number of reference sites per cluster (at least 30) and the largest number of clusters to increase our description of the spatial variability of fish faunas at a large scale. This procedure (Fig. 1) resulted in 11 clusters or river groups (RIVG).

CANDIDATE METRICS

We considered five functional attributes to define the list of candidate metrics: tolerance, trophic, reproduction, habitat and migration (Hughes & Oberdorff 1999; Oberdorff *et al*. 2001). The 309 fish species caught were assigned for these attributes, based on previous grey or published literature and completed by expert judgement when necessary (see the web site http://fame.boku.ac.at/publications.htm, 19 Dec 2005). Given the scale of the study and the number of species involved, natural history cannot be described in a rigorous, quantitative way for all the species. Nevertheless, our species assignments match well (average percent of match of 80%) with those realized recently for nine species attributes in Romania (Angermeier & Davideanu 2004).

Reproduction

Lithophilic species (LITH) require unsilted mineral substrate to spawn and their larvae are photophobic (Balon 1975). They tend to decrease in response to human disturbances such as siltation (Berkman & Rabeni 1987) and channelization (Brookes, Knight & Shields 1996). Phytophilic species (PHYT) tend to spawn on vegetation and their larvae are not photophobic. They decrease in response to channelization but will commonly increase with aquatic vegetation in relation with eutrophication.

Habitat

The water column (WATE), benthic (BENTH), rheophilic (RHEO) and limnophilic (LIMN) species prefer to live and feed in their respective habitat. The abundance of species assigned to these four habitat attributes tends to decrease with increasing habitat alteration (Karr 1981; Oberdorff *et al*. 2002). RHEO species may also increase when river channelization increases flow velocity. Eurytopic species (EURY) are characterized by tolerance of contrasting flow conditions, and an increase would be indicative of alteration.

Trophic guilds

Obligatorily piscivorous species (PISC; more than 75% fish in the diet; Lyons *et al*. 1995; Goldstein & Simon 1999) and insectivorous/invertivorous species (INSE; more than 75% macro-invertebrates in the diet; Lyons *et al*. 1995) will tend to decrease in response to an alteration of their habitat. In contrast, a metric based on omnivorous species (OMNI; more than 25% plant material and more than 25% animal material; Schlosser 1982) will tend to increase in response to disturbance as OMNI are able to adapt their trophic regime in response to an alteration of river food webs (Karr 1981).

Tolerance

Tolerant (TOLE) and intolerant (INTO) groups reflect species sensitivity to any common impact related to altered flow regime, nutrient regime, habitat structure and water chemistry (Karr *et al*. 1986). Loss of intolerant species is a response to degradation, whereas the number of tolerant species will tend to increase with disturbance.

Migration

Potamodromous species (POTA), which migrate within the inland waters of a river system (Northcote 1999), and long-migratory (diadromous) species (LONG), which migrate across a transition zone between fresh and marine water, are expected to decrease in response to the effects induced by dams and water regulation.

The choice of how a metric is expressed is as important as the selection of the metric itself (Fausch *et al*. 1990; Karr & Chu 1999). Each candidate metric was therefore expressed in four units: number of species

(Ns), relative number of species ($\%Ns$; number of species divided by the total species richness), absolute densities (Ni; in number of individuals ha^{-1}) and relative densities ($\%Ni$). Total species richness (RICH) and total abundance (DENS) would generally decline with environmental degradation (Karr 1981). However, an increase in nutrients (eutrophication) or temperature can also lead to an increase with disturbance.

We also considered metrics based on non-native acclimated species, as they can play an important functional role in river ecosystems. Finally, all 309 species caught were classified into one or several guilds for which we could calculate 58 candidate metrics.

METRIC MODELLING

For metrics based on abundance data ($n = 15$), stepwise multi-linear regression analysis of each metric [log ($x + 1$) transformed] on the explanatory variables was used. The squares of each of the five quantitative variables (ELE, TEMP, DIS, SLOP, WID) were also included to allow for non-linear relationships. For metrics based on the number of species ($n = 15$), we used the same procedure but added an explanatory variable, FISH, as sampling area is well known to influence species richness (Angermeier & Schlosser 1989). For metrics based on the relative number of species or relative abundance ($n = 28$), we used stepwise logistic regression analysis. All these analyses were performed using our calibration reference data set (REF-CAL). RIVG, TECH, METH, LAK, GEO and FLOW were entered as dummy variables. Variable selection during the stepwise procedure was based on the Akaike information criteria (Hastie & Pregibon 1993).

Using the independent set of 304 reference sites (REF-MET), we validated each of the resulting metric-specific models, expecting that the intercept and the slope of the regression line of the observed vs. predicted values would not be significantly different from, respectively, 0 and 1. In addition, we arbitrarily set a minimal threshold of the variance explained by the model (determination coefficient) at 0·30.

The residuals of each of the valid metrics (i.e. the deviation between the observed and the predicted value of a metric) measured the range of variation of metrics after eliminating the effects of environmental variables and in the absence of any human disturbance. These residuals were standardized through subtraction and division by, respectively, the mean and the standard deviation of the residuals of the reference calibration data set (REF-CAL), even when computed on other data sets (REF-MET, WI-MET, HI-MET, REF-IND, WI-IND, HI-IND).

Most of the metrics are expected to be negatively linked to the intensity of human perturbation. This means that the expected value of the residuals for reference sites is zero and less than zero for impacted sites. Assuming that standardized residuals are $N(0,1)$ distributed within reference sites, it is possible to compute the probability of observing a residual value lower than the computed one. The lower this probability, the higher the probability that a site is impacted. For metrics that are expected to be positively linked to human disturbance, we estimated the probability of observing values higher than the computed ones. For metrics that are expected to respond by an increase or a decrease depending on the type of perturbation, we considered the probability of observing higher values than the computed one for positive residuals, or of observing lower values for negative residuals. Transforming residual metrics into probabilities as described above is a way of rendering them comparable. All probability metrics vary between zero and one, and decrease as human disturbance increases. The expected distribution of these probabilities for reference sites is a uniform distribution with mean = 0·5.

METRIC SELECTION

Metrics were selected after validation with the REF-MET data set on the grounds of their sensitivity to human-induced disturbance, and to maximize the independence among metrics and the diversity of metric types. First each metric was calculated for the subsets of weakly (WI-MET) and heavily (HI-MET) altered sites to test the hypothesis that the mean probability value of REF-MET was higher than WI-MET, and that the mean probability value of WI-MET was higher than HI-MET (unilateral t-test). Only metrics that fulfilled these two criteria were retained hereafter.

Then if two metrics were highly correlated (i.e. Pearson's $r < 0·80$ or $> -0·80$), we retained the metric based on a functional species attribute not yet selected or the metric demonstrating the strongest response to a high level of disturbance.

INDEX CALCULATION AND VALIDATION

For a site, given its fish assemblage, geographical location, environmental features and the sampling method used, applying the models corresponding to each of the 10 metrics produces 10 residual values (one per metric) that are subsequently transformed into probabilities. The final index is obtained by summing up the 10 probabilities and rescaling the final score from 0 to 1 (by dividing it by 10). The index was validated on three new independent subsets: reference (REF-IND), weakly disturbed (WI-IND) and highly disturbed (HI-IND) sites. We hypothesized that (i) the mean value of REF-IND did not differ from 0·5 and (ii) REF-IND mean value > WI-IND mean value > HI-IND mean value (unilateral t-test).

Two explicit examples (Table 2) that convert actual site biological and environmental conditions into metric scores and a final index score are given. A software freely available on the web (http://fame.boku.ac.at) may be used for this purpose.

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Table 1. List of the 29 metrics retained after the first validation procedure of the multiple linear or logistic models. Expected metric responses to human disturbances: positive response (+), negative response (–), positive or negative response (+/–). R^2, determination coefficients of the regression of observed vs. predicted metric values using the independent reference data set (REF-MET). Mean metrics values (after standardization and transformation into probabilities; see text for detailed explanations) for REF-MET and the two data sets of weakly (WI-MET) and highly (HI-MET) disturbed sites. *P*-values of Student's *t*-test comparing REF-MET to WI-MET, and WI-MET to HI-MET

Metrics	Expected response	R^2	Mean value (REF-MET)	Mean value (WI-MET)	Mean value (HI-MET)	*P*-value (REF-WI)	*P*-value (WI-HI)
Ni-PISC	–	0·402	0·459	0·363	0·387	< 0·00001	0·95150
Ni-INSE	–	0·353	0·560	0·228	0·066	< 0·00001	< 0·00001
Ni-OMNI	+	0·407	0·512	0·417	0·365	< 0·00001	0.00040
Ni-EURY	+	0·461	0·465	0·431	0·434	0·04870	0·56920
Ni-LONG	–	0·383	0·509	0·293	0·208	< 0·00001	< 0·00001
Ni-LITH	–	0·308	0·548	0·258	0·125	< 0·00001	< 0·00001
Ni-PHYT	+	0·357	0·530	0·470	0·423	0·00290	0·00310
Ni-TOLE	+	0·446	0·516	0·466	0·479	0·00630	0·80870
Ns-PISC	–	0·466	0·461	0·391	0·406	0·00020	0·84140
RICH	+ –	0·608	0·481	0·351	0·339	< 0·00001	0·21240
Ns-OMNI	+	0·552	0·517	0·462	0·415	0·00530	0·00170
Ns-BENT	–	0·481	0·527	0·338	0·267	< 0·00001	< 0·00001
Ns-EURY	+	0·555	0·482	0·480	0·458	0·47120	0·07470
Ns-RHEO	–	0·423	0·512	0·243	0·101	< 0·00001	< 0·00001
Ns-WATE	–	0·550	0·500	0·417	0·397	0·00010	0·10560
Ns-LONG	–	0·353	0·504	0·275	0·212	< 0·00001	< 0·00001
Ns-POTA	–	0·439	0·499	0·404	0·308	< 0·00001	< 0·00001
Ns-LITH	–	0·398	0·512	0·230	0·070	< 0·00001	< 0·00001
Ns-TOLE	+	0·493	0·514	0·470	0·464	0·01980	0·34700
%*Ni*-EURY	+	0·423	0·473	0·537	0·543	0·99940	0·64010
%*Ni*-RHEO	–	0·326	0·529	0·426	0·296	< 0·00001	< 0·00001
%*Ni*-LONG	–	0·376	0·515	0·517	0·541	0·56200	0·99570
%*Ni*-LITH	–	0·402	0·528	0·386	0·244	< 0·00001	< 0·00001
%*Ni*-TOLE	+	0·478	0·519	0·502	0·418	0·20010	< 0·00001
%*Ns*-INSE	–	0·429	0·510	0·325	0·213	< 0·00001	< 0·00001
%*Ns*-EURY	+	0·346	0·457	0·607	0·585	1·00000	0·08360
%*Ns*-INTO	–	0·453	0·519	0·314	0·186	< 0·00001	< 0·00001
%*Ns*-LITH	–	0·389	0·532	0·260	0·097	< 0·00001	< 0·00001
%*Ns*-TOLE	+	0·307	0·538	0·325	0·286	< 0·00001	0·00650

Results

Twenty-nine of the 58 metrics were validated (Table 1). Regressions between observed and predicted values were highly significant (R^2 30·7–60·8%). The intercepts and the slopes of the corresponding regression lines did not significantly differ from zero (Student's *t*-test; *P*-values from 0·071 to 0·954) and one (*P*-values from 0·055 to 0·969), respectively. Residual distributions were checked graphically to verify that they were symmetrical with only a few outliers.

Residuals were calculated and standardized (deviation between the observed and the predicted value; Table 2) for each metric and for each of the three data sets REF-CAL, WI-MET and HI-MET. We then transformed these residuals into probabilities in agreement with our previously defined response hypotheses (Table 1). Among the remaining metrics, 17 demonstrated a significant difference between REF-CAL and WI-MET mean values (*P*-values from 0·003 to < 0·000001) and between WI-MET and HI-MET mean values (*P*-values from 0·007 to < 0·000001). At this step, three of the five habitat-types metrics (WATE, LIMN, EURY), PISC and RICH metrics were excluded. As expected, the REF-CAL mean values were very close to 0·5 (from 0·499 to 0·560) for all the retained metrics. The responses to degradation were in agreement with our previous hypotheses: OMNI, TOLE and PHYT metric types increased, while the seven other metric types decreased. But metric responses varied in intensity, with the weakest deviation for metric types demonstrating a positive response to human disturbance (OMNI, TOLE and PHYT).

Five of the 17 remaining metrics were strongly correlated (*Ni*-OMNI and *Ns*-OMNI, Pearson's coefficient R = 0·85; *Ni*-LONG and *Ns*-LONG, R = 0·89; *Ns*-RHEO and *Ns*-LITH, R = 0·97; %*Ni*-RHEO and %*Ni*-LITH, R = 0·81; %*Ns*-INSE and %*Ns*-INTO, R = 0·89). We finally retained 10 metrics (for regression coefficients see web site http://fame.boku.ac.at/publications.htm, 19 Dec 2005): two trophic-based metrics (*Ni*-INSE, *Ni*-OMNI), two reproductive guild-based metrics (%*Ns*-LITH, *Ni*-PHYT), two habitat-based metrics (*Ns*-BENT, *Ns*-RHEO), two migration status-based metrics (*Ns*-POTA, *Ni*-LONG) and two tolerance status-based metrics (%*Ns*-INTO, %*Ns*-TOLE). Three of these

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Table 2. Two examples of calculation of the European fish index value from the 10 retained metrics used in the model (site 1, undisturbed site with no individual disturbance variable rated over 1; site 2, highly disturbed site with a global disturbance value (DISTURB) of 14). For each site, the list of species caught (within parentheses, number of individuals caught per species at a given sampling date and the metric set to which the species was assigned) and environmental conditions (see text for acronym signification) are given. Metrics (see text for acronym signification) are expressed in number of species (*Ns*), relative number of species (%*Ns*; number of species divided by the total species richness), absolute densities (*Ni*; in number of individuals ha^{-1}) and relative densities (%*Ni*). Fish assemblage characteristics are converted into an observed metric. Environmental conditions are used to compute a theoretical metric value. The observed minus the predicted values are standardized and transformed into probabilities. In the absence of any disturbance, a value of 0·5 is expected. The index is obtained by summing up the 10 metrics

Site 1	*River Leven (UK)*
Sampling date	23 August 2002
Fish assemblage	*Anguilla anguilla* (1, TOLE, BENT, LONG), *Barbatula barbatula* (5, BENT, RHEO, LITH), *Cottus gobio* (100, INTO, BENT, RHEO, LITH, INSE), *Lampetra planeri* (1, INTO, BENT, RHEO, LITH, POTA), *Phoxinus phoxinus* (55, RHEO, LITH), *Salmo salar* (53, INTO, RHEO, LITH, INSE, LONG), *Salmo trutta fario* (39, INTO, RHEO, LITH, INSE)
Environmental conditions	RIVG (United.Kingdom), CAT (< 100 km^2), ELE (45 m), GEO (Siliceous), FLOW (Permanent), LAK (No), TEMP (8·5 °C), SLOP (2·75 m km^{-1}), DIS (14 km) WID (7·3 m), METH (Whole), TECH (Wading), FISH (365 m^2)
Index value	0·80

Metrics	*Ni*-INSE*	*Ni*-OMNI*	*Ni*-PHYT*	*Ns*-BENT*	*Ns*-RHEO*	*Ns*-LONG*	*Ns*-POTA*	% *Ni*-LITH	% *Ns*-INTO	% *Ns*-TOLE
Observed	8·568	0·000	0·000	1·609	1·946	1·099	0·693	0·996	0·570	0·140
Predicted	6·847	2·717	0·788	1·092	1·506	0·706	0·227	0·009	0·004	0·002
Probability	0·841	0·896	0·723	0·896	0·889	0·905	0·880	0·727	0·645	0·636

Site 2	*Seine River (FR)*
Sampling date	19 September 1996
Fish assemblage	*Abramis brama* (4, TOLE, BENT, OMNI, POTA), *Anguilla anguilla* (22, TOLE, BENT, LONG), *Gobio gobio* (5, INTO, BENT, RHEO, LITH, INSE), *Leuciscus cephalus* (13, RHEO, LITH, OMNI, POTA), *Perca fluviatilis* (6, TOLE), *Rutilus rutilus* (159, TOLE, OMNI), *Sander lucioperca* (1), *Scardinius erythrophthalmus* (6, PHYT, OMNI)
Environmental conditions	RIVG (West.France), CAT(> 10000 km^2), ELE (8 m), GEO (Calcareous), FLOW (Permanent), LAK (No), TEMP 10·5 °C, SLOP (1·0 m km^{-1}), DIS (615 km) WID (100 m), METH (Partial), TECH (Boat), FISH (1440 m^2)
Index value	0·16

Metrics	*Ni*-INSE	*Ni*-OMNI	NI-PHYT	*Ns*-BENT	*Ns*-RHEO	*Ns*-LONG	*Ns*-POTA	% *Ni*-LITH	*Ns*-INTO	% *Ns*-TOLE
Observed values	0·000	7·143	3·761	1·386	1·099	0·693	1·099	0·060	0·000	0·500
Predicted values	5·481	3·554	0·961	1·929	2·069	1·066	0·894	0·236	0·184	0·261
Probability	0·001	0·048	0·018	0·093	0·004	0·107	0·698	0·189	0·159	0·037

*Metrics expressed in ln(x + 1).

metrics responded positively to human disturbance (*Ni*-OMNI, *Ni*-PHYT, %*Ns*-TOLE).

The metric values and final index score were computed for the three independent data sets (REF-IND, WI-IND, HI-IND) (Fig. 2). The mean value of the index (0·513) in the reference data set did not differ significantly from 0·5 (t = 1·834, P = 0·067). The mean index value (0·343) in the weakly disturbed sites (WI-IND) was significantly lower than that of the reference sites (t = 16·546, P < 0·000001) and significantly higher than that of the highly disturbed sites (0·235, t = 10·36, P < 0000001).

By examining the percentages of well-classified reference (REF-IND) and disturbed sites (WI-IND and HI-IND) as a function of each index score value, we demonstrated that the best cut-off level for assemblage 'impairment' was an index value of 0·423, with 81·4% of the reference and disturbed sites correctly classified.

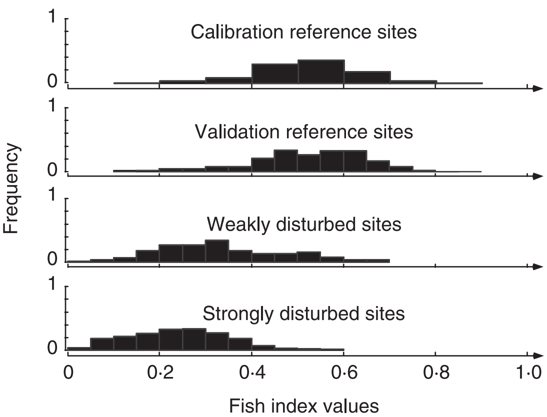

Fig. 2. Distribution of the index scores for REF-CAL (calibration reference sites), REF-IND (independent reference sites, n = 304), WI-IND (weakly disturbed sites, n = 304) and HI-IND (highly disturbed sites, n = 304).

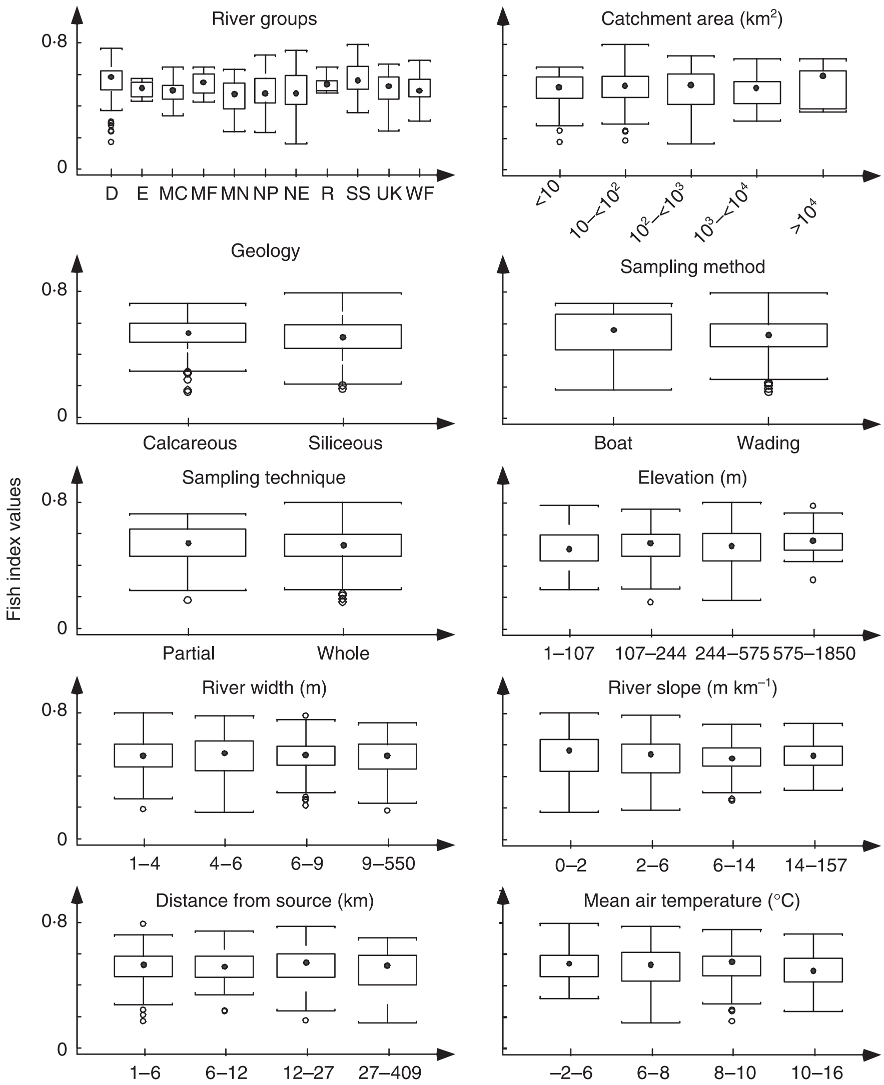

Fig. 3. Distribution of the index score for each of the 10 environmental variables (box-plot graphs) for the two independent validation data sets (REF-MET and REF-IND, $n = 608$).

In order to test index independence to natural environmental variability, we performed a stepwise linear regression of the index values on all 10 environmental descriptors, using the independent reference data set (REF-IND). None of the descriptors was retained and the part of the index variability explained by these descriptors was not significant ($R^2 = 0{\cdot}115$, F-test = 1·505, $P = 0{\cdot}064$). When considering each of the 10 environmental descriptor separately (Fig. 3), multiple comparison Tukey's test showed that the index values were invariant, whatever the value of the descriptor tested, except for the two highest elevation classes.

To evaluate the ability of any impacted site to deviate from a reference condition (i.e. a mean index value of 0·5), we regressed the index values of all independent sites (REF-IND, WI-IND, HI-IND) on the global assessment impact variable (IMPACT). The relationship (Fig. 4) was highly significant ($R^2 = 0{\cdot}4678$, $n = 912$, $P < 0{\cdot}000001$). The standard error was 0·1236 and the residual distribution was normalized (goodness-of-fit Kolmogorov–Smirnov test, $P = 0{\cdot}6013$). Residuals

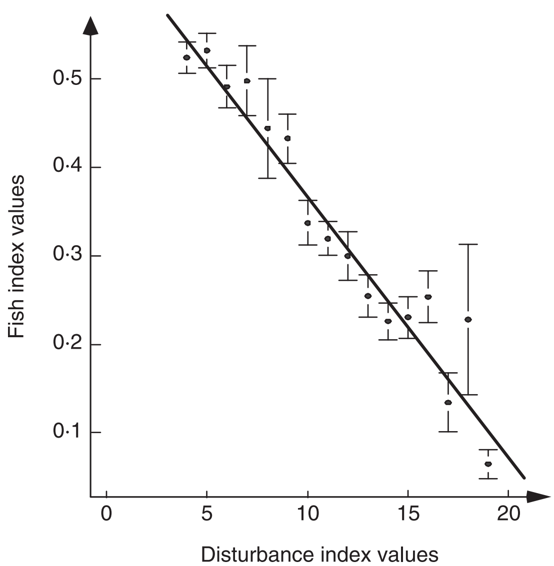

Fig. 4. Regression of the fish index values on the total human disturbance index (from four to 19). Mean index values (and their 95% confidence intervals) for each disturbance index class.

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at disturbance level 18 were highly dispersed because of the presence of an outlier (index value of 0·44) and because of the low number of sites (seven) at this level.

Discussion

This study demonstrates that at a continental-scale (Europe) it was possible to develop a multimetric fish-based index that (i) remained invariant for all unimpaired sites, whatever the natural environmental conditions, and (ii) gave a significant negative linear response to a gradient of physical and chemical human disturbances (i.e. the index can be used to assess the human-induced impact on the biotic condition in rivers). To our knowledge, this is the first time this kind of index has been successfully developed at such a large spatial scale. This is particularly noteworthy given the great variability of fish composition across different European biogeographical areas. Three features of our approach may have contributed to this. First, using functional metrics instead of taxonomic metrics reduced the sensitivity of the index to the variability in fish faunas between biogeographical regions. Secondly, including the main factors known to affect fish assemblage structure in the model reduced the influence of the geographical and upstream–downstream variability of these variables. And finally, including a biologically based regionalized variable added spatial flexibility to our approach.

The advantage of considering functional metrics can be illustrated by the case of lithophilic species. Among the 63 lithophilic species included in the LITH metric type at the European scale, only two species are common to all 11 hydrological units (brown trout and rainbow trout). Only 23 species occur in six units while 17 species are only present in one hydrological unit. Despite this variability in species composition between river units, LITH consistently decreased in response to human disturbance across Europe as a whole, demonstrating that the retained metrics are truly functional ones.

As demonstrated by our models, functional descriptors of fish communities respond to environmental variability in several ways. But all the 10 retained metrics responded significantly to river slope. Sampling methods also significantly affected eight of the 10 metrics, as demonstrated previously by Reynolds *et al*. (2003).

A tenet of our approach is that the variance of the metrics not accounted for by the environmental variables included in the models should be in large part the result of human disturbance. In fact, human disturbance only explains about 50% of total index variance, suggesting that the models may be improved by adding environmental variables not considered in this study. The unexplained variance in the index may also result from imprecision in fish sampling because of the inescapable differences in fish sampling methods used between different habitats and countries. Data on different types of river modifications were not always comparable between countries, so we only retained the four most reliable and complete disturbance variables. However, others types of disturbance have to be considered, such as fishing and introduced species, which can affect biotic interactions.

Moreover, riverine fish assemblages may vary greatly over time. To check this, the variance in index scores associated with the temporal variability of fish assemblages and/or sampling variability was evaluated by computing the standard deviation associated with 12 time series (eight to 36 sampling dates) distributed among four countries (Belgium, France, Lithuania, Sweden), at sites where there were no perceivable changes in human disturbance intensity during the period sampled. A mean of the standard deviations per site of 0·06 can be compared with the value of 0·169 observed for references sites. As the variance of the index within reference sites is the result of non-modelled spatial variability, as well as to sampling noise and temporal variability, this suggests that sampling noise accounts for at most 35% of index variability. Hence the most probable solution to improving the power of the index would appear to be by improving the modelling of its spatial variability (e.g. by considering new variables or other modelling approaches).

Another predictive approach based on the modelling of assemblages within reference sites as a function of environmental variables, RIVPACS, has recently been applied to fish assemblages in New Zealand (Joy & Death 2002). Besides the way assemblages are modelled (classification vs. regression), the main difference with our approach is the use of taxonomic richness instead of several metrics. In a sense our metrics RICH may be considered a RIVPACS-type descriptor inserted within a more general, multi-metrics, approach. As a result we expect our approach to be more powerful and more flexible. Despite the fact that we modelled functional metrics instead of taxonomic metrics and that we included some important environmental variables, the spatial variability in metric values has not been fully accounted for by our models, as exemplified by the inclusion of the regionalized variable 'river group' in eight of the 10 retained models. The two metrics types that are insensitive to regional classification of fish fauna are the omnivorous species (*Ni*-OMNI) and tolerant species (%*Ns*-TOLE), i.e. metrics comprised of generalist species able to colonize a wide spectrum of river environments. Among the 29 characteristically omnivorous species, 21 of them are common to at least six of the 11 river groups.

In previous works, most authors consider that assessment criteria must be region-specific (Angermeier, Smogor & Stauffer 2000). Ecologically defined regions are thereby considered to be relevant entities even if there is still considerable debate regarding how they should be defined (Omernik & Bailey 1997; Van Sickle & Hughes 2000). In our approach, we explicitly considered this question by including in our models an environmental variable acting at regional scales (river groups), in accordance with current views emphasizing regional influences on biodiversity (Ricklefs & Schluter 1993). Our regional classification is generally in agreement with the classical biogeographical history of Europe (Banarescu 1992). Fish faunas from the Netherlands, northern Germany, Poland, Lithuania and northern Sweden (European North Plain river group) appear as similar, related to their common recolonization after the glacial periods. The Baltic Sea was oligohaline and did not represent an ecological barrier to dispersal. The 'river group' variable participates additively in our models, meaning that the models are qualitatively consistent over Europe. For instance, a metric that is positively correlated to river slope in a given region will also be positively correlated with river slope in other European regions. Hence the models are partly transferable between regions but some regional adjustments are needed. Interregional variations may be linked to variation in taxonomy and phylogenetic history that in turn affect metric distribution within faunas, and also to spatial variation in environmental constraints not included into models but which can also affect functional characteristics of fish assemblages (Smogor & Angermeier 2001). Clearly further work is needed to identify the factors underlying the regional component of our models.

This present approach is based on modelling fish assemblage structure in reference sites. Thus the definition and quality of the reference data set are key issues. We collected information from a very large number of sites distributed among 1843 European rivers, compiling a database unique in Europe. Although sites are not evenly distributed, they cover virtually all the environmental situations a European fish species can encounter within its area of distribution. However, natural large flood plain rivers were lacking in our reference data set, because of their rarity in western Europe, and the efficiency of our index to assess such environments needs to be improved in the future. We did not restrict our selection of reference sites to only undisturbed sites but also considered sites slightly impacted. However, the mean index value for undisturbed sites is slightly but significantly higher than for slightly impacted sites ($5 \leq$ DISTURB ≤ 8) (0·518 against 0·502; t-test value = 2·349, $P = 0{\cdot}019$), suggesting that distinguishing between these two groups in the future would increase the power of the index.

In conclusion, the need to define sensitive biological measures of aquatic ecosystem integrity transferable to other catchments or regions at continental scales is now clear, especially in Europe with the implementation of the WFD. The solution we have used to meet this goal is to include in our reference condition modelling approach a more complete description of abiotic and biotic environmental variability at both local and regional scales. This sort of tool has never been developed before at a continental scale. Using this approach, our models and the final index are transferable, but only for sites and rivers belonging to the area considered in our previous calibration data set (REF-CAL). A generalization of our method to an even larger area is possible but only by collecting new data covering these new areas and by recalibrating our models. This methodology could also be improved by including better biological knowledge in the definition of the metric types, improving disturbance assessment and by using new statistical techniques. Lastly, the principles of our methodology could be applied to a wide variety of biological groups.

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References

Angermeier, P.L. & Davideanu, G. (2004) Using fish communities to assess streams in Romania: initial development of an index of biotic integrity. *Hydrobiologia*, **511**, 65–78.

Angermeier, P.L. & Schlosser, I.J. (1987) Assessing biotic integrity of the fish community in a small Illinois stream. *North American Journal of Fisheries Management*, **7**, 331–338.

Angermeier, P.L. & Schlosser, I.J. (1989) Species–area relationships for stream fishes. *Ecology*, **70**, 1450–1462.

Angermeier, P.L., Smogor, R.A. & Stauffer, J.R. (2000) Regional frameworks and candidate metrics for assessing biotic integrity in mid-atlantic highland streams. *Transactions of the American Fisheries Society*, **129**, 962–981.

Bailey, R.C., Kennedy, M.G., Dervish, M.Z. & Taylor, R.M. (1998) Biological assessment of freshwater ecosystems using a reference approach: comparing predicted and actual benthic invertebrate communities in Yukon streams. *Freshwater Biology*, **39**, 765–774.

Balon, E.K. (1975) Reproductive guilds of fishes: a proposal and definition. *Canadian Journal of Fisheries and Aquatic Sciences*, **32**, 821–864.

Banarescu, P. (1992) *Zoogeography of Fresh Waters. II. Distribution and Dispersal of Freshwater Animals in North America and Eurasia.* AULA-Verlag, Wiesbaden, Germany.

Berkman, H.E. & Rabeni, C.F. (1987) Effect of siltation on fish communities. *Environmental Biology of Fishes*, **18**, 285–294.

Brookes, A., Knight, S.S. & Shields, D. Jr (1996) Habitat enhancement. *River Channel Restoration* (eds A. Brookes & F.D. Shields, Jr), pp. 103–126. John Wiley & Sons Ltd, Chichester, UK.

Emery, E.B., Simon, T.P., McCormick, F.H., Angermeier, P.L., Deshon, J.E., Yoder, C.O., Sanders, R.E., Pearson, W.D., Hickman, G.D., Reash, R.J. & Thomas, J.A. (2003) Development of a multimetric index for assessing the biological condition of the Ohio River. *Transactions of the American Fisheries Society*, **132**, 791–808.

Fausch, K.D., Lyons, J., Karr, J.R. & Angermeier, P.L. (1990) Fish communities as indicators of environmental degradation. *American Fisheries Society Symposium*, **8**, 123–144.

Giller, P.S. (2005) River restoration: seeking the ecological standards. *Journal of Applied Ecology*, **42**, 201–207.

Goldstein, R.M. & Simon, T.P. (1999) Toward a united definition of guild structure for feeding ecology of North American freshwater fishes. *Assessing the Sustainability and Biological Integrity of Water Resources using Fish Communities* (ed. T.P. Simon), pp. 123–220. Lewis Press, Boca Raton, FL.

Harris, J.H. & Silveira, R. (1999) Large-scale assessments of river health using an index of biotic integrity with low-diversity fish communities. *Freshwater Biology*, **41**, 235–252.

Hastie, T.J. & Pregibon, D. (1993) Generalized linear models. *Statistical Models in S* (eds J.M. Chambers & T.J. Hastie), pp. 195–247. Chapman & Hall, London, UK.

Hughes, R.M. & Oberdorff, T. (1999) Applications of IBI concepts and metrics to waters outside the United States and Canada. *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities* (ed. T.P. Simon), pp. 79–83. Lewis Press, Boca Raton, FL.

Hughes, R.M., Howlin, S. & Kaufmann, P.R. (2004) A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. *Transactions of the American Fisheries Society*, **133**, 1497–1515.

Hughes, R.M., Kaufmann, P.R., Herlihy, A.T., Kincaid, T.M., Reynolds, L. & Larsen, D.P. (1998) A process for developing and evaluating indices of fish assemblage integrity. *Canadian Journal of Fisheries and Aquatic Sciences*, **55**, 1618–1631.

Hughes, R.M., Whittier, C.M.R. & Larsen, D.P. (1990) A regional framework for establishing recovery criteria. *Environmental Management*, **14**, 673–683.

Joy, M.K. & Death, R.G. (2002) Predictive modelling of freshwater fish as a biomonitoring tool in New Zealand. *Freshwater Biology*, **47**, 2261–2275.

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

Karr, J.R. & Chu, E.W. (1999) *Restoring Life in Running Waters: Better Biological Monitoring*. Island Press, Washington, DC.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R. & Schlosser, I.J. (1986) *Assessing Biological Integrity in Running Waters: A Method and its Rationale*. Special Publication No. 5. Illinois Natural History Survey, Champaign, IL.

Leonard, P.M. & Orth, D.J. (1986) Application and testing of an index of biotic integrity in small, coolwater streams. *Transactions of the American Fisheries Society*, **115**, 401–414.

Lyons, J., Navarro-Perez, S., Cochran, P.A., Santana, E.C. & Guzman-Arroyo, M. (1995) Index of biotic integrity based on fish assemblages for the conservation of streams and rivers in West-Central Mexico. *Conservation Biology*, **9**, 569–584.

McCormick, F.H., Hughes, R.M., Kaufmann, P.R., Peck, D.V., Stoddard, J.L. & Herlihy, A.T. (2001) Development of an index of biotic integrity for the mid-Atlantic highlands region. *Transactions of the American Fisheries Society*, **130**, 857–877.

Matthews, W.J. (1998) *Patterns in Freshwater Fish Ecology*. Chapman & Hall, New York, NY.

Mebane, C.A., Maret, T.R. & Hughes, R.M. (2003) An index of biological integrity (IBI) for Pacific Northwest rivers. *Transactions of the American Fisheries Society*, **132**, 239–261.

Northcote, T.G. (1999) Migratory behaviour of fish and its significance to movement through riverine fish passage facilities. *Fish Migration and Fish Bypasses* (eds S. Jungwirth, S. Schmutz & S. Weiss), pp. 3–18. Fishing News Books, Blackwell Science, Oxford, UK.

Oberdorff, T., Pont, D., Hugueny, B. & Chessel, D. (2001) A probabilistic model characterizing fish assemblages of French rivers: a framework for environmental assessment. *Freshwater Biology*, **46**, 399–415.

Oberdorff, T., Pont, D., Hugueny, B. & Porcher, J.P. (2002) Development and validation of a fish-based index (FBI) for the assessment of 'river health' in France. *Freshwater Biology*, **47**, 1720–1734.

Omernik, J.M. & Bailey, R.G. (1997) Distinguishing between watersheds and ecoregions. *American Water Resource Association*, **33**, 935–949.

Ormerod, S.J. (2003) Current issues with fish and fisheries: editor's overview and introduction. *Journal of Applied Ecology*, **40**, 204–213.

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

Pont, D., Hugueny, B. & Oberdorff, T. (2005) Modelling habitat requirement of European fishes: do species have similar responses to local and regional environmental constraints? *Canadian Journal of Fisheries and Aquatic Sciences*, **62**, 163–173.

Pretty, J.L., Harrison, S.S.C., Shepherd, D.J., Smith, A.G. & Hey, R.D. (2003) River rehabilitation and fish populations: assessing the benefit of instream structures. *Journal of Applied Ecology*, **40**, 251–265.

Reynolds, L., Herlihy, A.T., Kaufmann, P.R., Gregory, S.V. & Hughes, R.M. (2003) Electrofishing effort requirements for assessing species richness and biotic integrity in western Oregon streams. *North American Journal of Fisheries Management*, **23**, 450–461.

Ricklefs, R.E. & Schluter, D. (1993) *Species Diversity in Ecological Communities*. University of Chicago Press, Chicago, IL.

Roth, N.E., Southerland, M., Chaillou, J., Klauda, R., Kazyak, P., Stranko, S., Weisberg, S., Hall, J.R. & Morgan, R. (1998) Maryland biological stream survey: development of a fish index of biotic integrity. *Environmental Monitoring and Assessment*, **51**, 89–106.

Schleiger, S.L. (2000) Use of an index of biotic integrity to detect effects of land uses on stream fish communities in western–central Georgia. *Transactions of the American Fisheries Society*, **129**, 1118–1133.

Schlosser, I.J. (1982) Fish community structure and function along two habitat gradients in a headwater stream. *Ecological Monographs*, **52**, 395–414.

Simon, T.P. (1999) *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. Lewis Publishers, Boca Raton, FL.

Simon, T.P. & Emery, E.B. (1995) Modification and assessment of an index of biotic integrity to quantify water resource quality in great rivers. *Regulated Rivers: Research and Management*, **11**, 283–298.

Smogor, R.A. & Angermeier, P.L. (2001) Determining a regional framework for assessing biotic integrity of Virginia Streams. *Transactions of the American Fisheries Society*, **130**, 18–35.

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Classification and assessment of degradation in European running waters

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Abstract A pan-European, classification of the extent of environmental degradation from chemical, physical and biological pressures on fish communities as a precursor to assess the ecological status of running waters based on fish is proposed. Twenty-four potential pressures acting on fish communities at three different spatial scales (river basin, segment and site) were identified and class boundaries for high, good, moderate, poor and bad status, based on existing data and/or expert judgement, were defined. Four pressures (hydrological regime, morphological conditions, toxic or acid conditions, nutrients and organic load) were found to describe the majority of degradation at a specific site and these were combined into a single pressure variable to describe impact at each location. Principal Component Analysis showed that the four variables were correlated with other physical and chemical variables not included in the combined pressure variable. However, biological pressures, e.g. introduction of fish, and longitudinal connectivity were not well correlated, suggesting that two dimensions of human impact on stream fish were poorly accounted for. Low-resolution Geographical Information Systems (GIS) data (1 km grid) on land use and population density correlated well with the four chosen pressures, suggesting it is possible to use standardised GIS data to aid pre-classification of stream degradation.

KEYWORDS: ecological status, Europe, fish, Geographical Information System, pressures, stream degradation.

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Introduction

A crucial requirement for the assessment of ecological status of aquatic systems, as required under the European Union Water Framework Directive (WFD; 2000/60/EEC) is identification of reference conditions (Karr 1981). These are the set of conditions expected under minimal or no anthropogenic disturbance. To develop an objective assessment system, the impact of different human activities on ecological status of different water bodies is required. Although the importance of such a classification systems is recognised, no robust system focussing on fish is available on a continental scale.

Human interference on streams falls into three major categories: physical, biological and chemical. A human activity that has a direct effect is called a pressure, e.g. construction of a dam or agricultural development in the watershed. The environmental effect of the pressure is an impact, e.g. loss of fish migration pathways or increased eutrophication (Impress Group 2003). To establish a pre-classification system there is a need to: (i) list appropriate pressures acting on fish; (2) identify key pressures; (3) define class boundaries separating no degradation and severe deviation for each pressure; (4) quantify each pressure for each water body; and (5) derive, from the quantified pressures, a single variable describing the degradation of each water body.

The objective of this paper was to identify key pressures that act on fish community and population structure, establish class boundaries for the scale of impact of these pressures on fish, and construct a single pressure variable. An overview of the spatial scale and intensity of these human pressures in European rivers is also provided.

Methods

Relevant pressure variables

A list of potential pressures was compiled from ANNEX V 1.2.1 of the WFD, the Impress Group (2003) documentation from the AQEM Consortium (2002) and the review by Cowx (2002). The compilation took into account that fish and macroinvertebrates may react differently to the same pressure (Paller 2001) that various kind of pressures act in different continental regions (cf. Barbieri, Economou, Stoumboudi & Economidis 2002; Freyhof 2002), and that the same pressure may have different impacts in different regions (Whittier & Hughes 1998).

Each potential pressure was reviewed by a panel of experts representing both governmental and scientific institutions from 12 European countries. It was concluded that for pressures and impacts to be included, they should have scientifically proven, potentially large, negative effects on fish, be relevant on a continental scale, act in the same direction with respect to stream degradation on a continental scale, and be possible to measure and quantify from existing data. It was also concluded that pressures should be classified in accordance with the WFD: i.e. high status (1); good status (2); moderate status (3); poor status (4); and bad status (5). After final review, 24 pressures were selected for inclusion (Table 1).

Impacts were assessed at three scales: river basin, segment and site. The site level was the area sampled by electric fishing, and averaged 177 m of river length. The length of the segment was set from catchment size and was 1 km for small rivers (catchment size: < 100 km^2), 5 km for medium-sized rivers (100–1000 km^2) and 10 km for larger rivers. Only physical impacts and global pressures (land use and urbanisation) were included at the river basin and segment scales, i.e. chemical and biological pressures were only accounted for at the site level (Table 1).

Class boundaries of pressures

Expert judgement, in conjunction with established criteria, e.g. classification of fish according to their tolerance to water quality (e.g. oxygen – Alabaster & Lloyd 1982; pH – Degerman & Lingdell 1993), were used to define class boundaries for individual pressures between different levels of degradation (Table 1). For example, for oxygen the classification was from lowest measured values: > 7mg L^{-1} or oxygen saturation $>$ 90% = class 1; < 7 mg L^{-1} or saturation 80–90% = class 2; < 5 mg L^{-1} or saturation 80–90% = class 3; 2–5 mg L^{-1} or saturation 70–80% = class 4; < 2 mg L^{-1} or saturation < 70% = class 5. Deviation from background levels was used for nutrients (phosphorous and nitrogen) and total organic carbon (TOC; Table 1).

For Geographical Information System (GIS) data, land use levels were obtained from the Impress Group (2003) guidance document. Connectivity was stressed by the WFD within the river continuity or ecological continuum concepts concerning hydromorphological quality. It was primarily included at the river basin and the segment scales, expressed as man-made migration barriers within the stream and their effects on fish migration. In high status (= 1) waters, no artificial barriers occurred or fully functional bypass facilities or similar devices were present, allowing all species and sizes to migrate. In waters

Table 1. The 24 human pressure and impact variables, the scale they apply to, abbreviated labels and a short explanation of classification. Type denotes if the variables describes Physical (P), Biological (B) or Chemical (C) impact or is regarded as a global pressure (GP)

Human impact variable	Scale	Type	Label	Explanation; short description of classes (1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad)
Connectivity, multiscale	River, segment	P	P.con.ms	See text
Connectivity, river	River	P	P.con.riv	See text
Land use	River	GP	G.landu.riv	Extent and impact of agriculture and silviculture; 1 = <10% of catchment area, 2 = <40% and low impact, 3 = <40% and moderate impact, 4 = >40% and strong impact, 5 = >40% and severe impact.
Urbanisation	River	GP	G.urb.riv	Extent and impact of urban areas; 1 = <1% of catchment area, 2 = <15% and low impact, 3 = <15% and moderate impact, 4 = >15% and strong impact, 5 = >15% and severe impact
Connectivity, segment	Segment	P	P.con.seg	See text
Land use	Segment	GP	G.landu.seg	Extent and impact of agriculture and silviculture; see above
Urbanisation	Segment	GP	G.urb.seg	Extent and impact of urban areas, see above
Riparian zone	Segment	P	P.ripz.seg	Deviation from natural state; 30–50 m perpendicular from shoreline, both banks; 1 = >90% in natural state, 2 = <90%, 3 = <75%, 4 = <50%, 5 = <25% in natural state
Floodplain lateral movements	Segment	P	P.lmov.seg	Floodplain with diverse water body types allowing lateral movements of fish; 1 = >90% in natural state, 2 = >50%, 3 = <25%, 4 = 10%, 5 = no floodplain left
Sediment load	Segment	P	P.sload.seg	Deviation from natural sediment load. Expert judgement
Hydrological regime	Site	P	P.hydr	Deviation from natural conditions, both flow pattern and quantity (worst of the two below)
Natural flow pattern	Site	P	P.natflowp	Deviation from natural flow pattern (water level and periodicity); 1 = >90% of level and duration of flood, 2 = >75%, 3 = >50%, 4 = <50%, 5 = <50% and strong deviation from natural
Natural flow quantity	Site	P	P.natflowq	Deviation from natural flow quantity (mean annual discharge, MQ); 1 = >90% of MQ, 2 = >30%, 3 = >15% of MQ, 4 = >15%, 5 = <10% of MQ
Upstream dam affects site	Site	GP	G.updam	Artificial upstream water body that affects temperature regime. Expert judgement
Morphological condition	Site	P	P.morph	Morphological alteration, e.g. channel form, reduction of habitats, in-stream features. 1 = negligible alteration, 2 = all habitat types present, 3 = channelised, some habitat types missing, 4 = channelised, most natural habitat types missing, 5 = canal
Salinity	Site	C	C.sal	Salinity, deviation from natural state; 1 = within normal variation, 3 = occasional deviation, 5 = long periods of strong deviation
Toxic or acid effects	Site	C	C.toxic	pH, oxygen and toxic compounds. For pH: pH > 6 = class 1; a single pH record of 5.5–6.0 = class 2; single pH of 5.0–5.5 = class 3; frequent pH < 5.5 = class 4, frequent pH < 5.0 = class 5. For oxygen see text
Nutrients and organic input	Site	C	C.nutr	Conditions within 150% of established natural background levels = high status. Sites with occasional small deviations (within the boundaries of the next class) = good status. Conditions within 150–300% of national background = moderate status. Occasional deviations above this = poor conditions and frequent or permanent deviations above 300% = bad status.
Introduction of fish	Site	B	B.intro	New fish species to river basin; 1 = no introduction, 2 = introduction, but no reproduction and low density, 3 = not reproduction, high density, 4 = reproducing, low density, 5 = reproducing, high density
Impact of fish stocking	Site	B	B.stock	Species already present, but otherwise as above (i.e. as B.intro)
Impact of exploitation	Site	B	B.expl	Human exploitation, e.g. fishing, at site affecting fauna. Expert judgement
Impact from other fauna	Site	B	B.fauna	Introduced, invasive or rapidly increasing species (not fish). Expert judgement
Impact from flora	Site	B	B.flora	Unnatural increase of helophytes and submerged plants. Expert judgement
Weed cutting	Site	B	B.weed	1 = never, 2 = >5 years ago, 3 = within 5 years, 4 = most years, 5 = several times a year

of good status (=2), passage over an artificial obstruction was possible for most species in most years. When migration was possible during certain years only or for only certain species, the impact was classified as moderate (=3). Poor status (=4) was when only single species could pass occasionally, and bad status when no species could pass (=5). To be able to assess the additive effects of connectivity at the river basin and segment scales, a combined variable, 'Connectivity at the multi-scale level', was calculated as the sum of connectivity on the river level (three levels) and connectivity on the segment level (two levels).

Introduction of new fish species to the river basin was classified on the basis of reproductive success and population density (Table 1). Most of the biological impact variables needed to be assessed by expert judgement as quantitative data were often missing.

Quantification of each pressure at each site and fishing occasion

Pressures at each site were quantified using environmental data previously gathered during fisheries surveys, data from maps and GIS, water chemistry data and expert judgement. There was considerable variability among countries over how data on the 24 pressures were gathered and processed. Data on hydrological regime, morphological conditions and the status of the riparian zone were included in most electric fishing field protocols but needed re-classification according to the prescribed boundaries. Geographical Information System was suggested as a possible source for eight variables but was used only for 35% of cases; instead 43% of classifications (all variables, all countries) were based on expert judgement but this incorporated best available information (e.g. historical data, local authorities, extrapolation from nearby sites and maps).

Key pressures and the single impact variable

As a first step in the assessment procedure, it is necessary to identify the key pressures that influence fish community structure and functioning. Data were available for 15 183 fishing occasions (8227 sites), but complete reporting of all impact variables was only available for 4067 occasions (870 sites). Thus, the number of variables that could be used to produce a joint single pressure variable was limited. Only seven pressure variables (C.toxic, P.morph, C.nutr., P.con.riv, P.con.seg, P.con.ms and P.hydr) were available for >80% of the sites, rising to 15 variables for 60% of sites (Fig. 1). Connectivity was represented by three different variables, but because of difficulties in scoring these variables in some countries, connectivity variables was found not to produce strong fish metric responses, thus connectivity was excluded. The combined pressure variable was therefore calculated as the average of the summed impact (1–5) of four variables (the core set variables); hydrological regime, morphological condition, toxic and acid compounds, nutrient and organic loading. Thus, degradation was accounted for only at the site scale, while connectivity and biological pressures were excluded.

Statistical analysis and comparisons with global GIS data

Principal Component Analysis (PCA) or centred Principal Component Analysis (cPCA) were used to evaluate correlations between pressures and scales. In the graphical presentation of the PCA only the first two dimensions are presented. The angle between the variables represents the correlation; a short arrow means that this particular variable is not well represented in the first two dimensions and that the correlation with the other variables was low.

Additional data on human population density (Landscan 2003) and land use (from Stockholm Environment Institute) were obtained from 1-km resolution GIS satellite imagery distributed by National Ocean and Atmospheric Administration. These data were included to determine if GIS-derived data correlate with the pressure variables could serve to interpret global pressure patterns. To compensate for possible positioning error, local neighbourhood values (mean or dominant) of the 1-km resolution layers were calculated for each site. These variables can therefore be seen as segment scale descriptors. The land use classification includes five categories of dominant land use types: semi-natural, forest, prairie, crops and urban. Landscan modelled population estimations were $\log_{10}$-transformed ($\log X + 1$) to normalise distributions and to improve homogeneity in the magnitude of their variances in relation to the other variables.

Results

Correlation between impact variables

Principal Component Analysis analyses were carried out to assess the contribution and redundancy of the variables not included against the four core pressure variables (Figs 2 and 3). The PCAs with the four

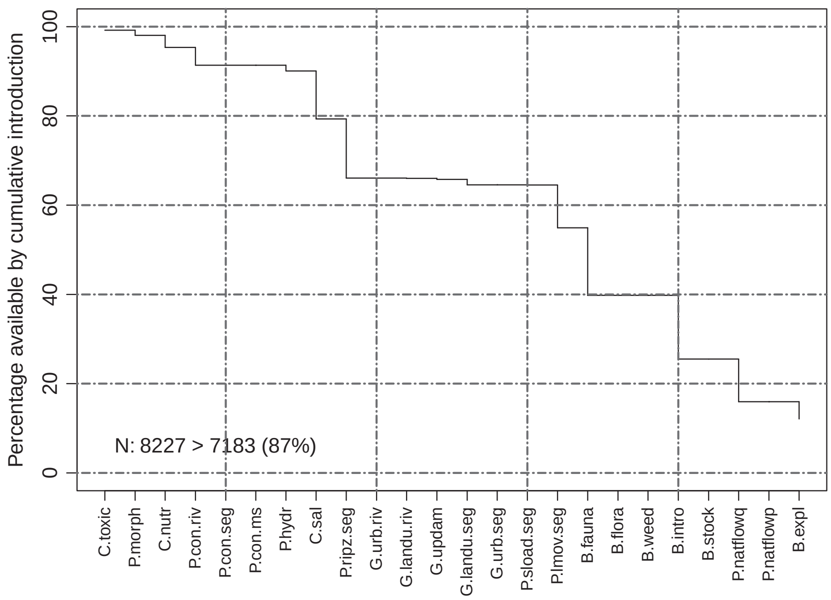

Figure 1. Percentage of sites that were available to multivariate analysis with respect to included variables describing human impact. The introduction of the variables is carried out in an optimal way. (See Table 1 for variable abbreviations.).

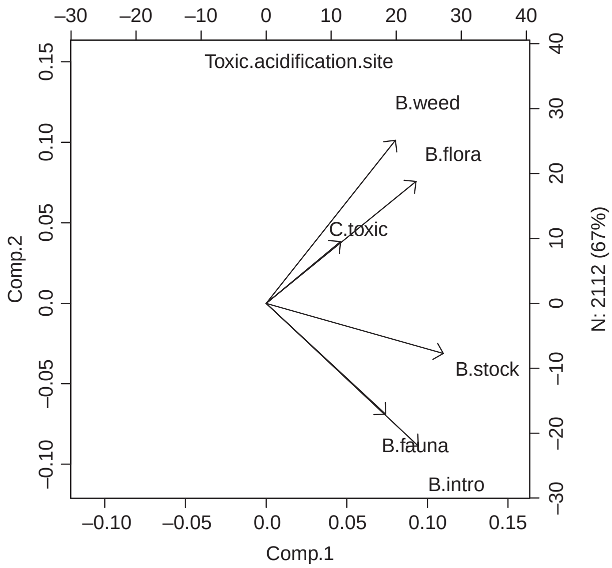

Figure 2. Principal Component Analysis with five biological variables and the variable 'toxic/acidification' (C.toxic).

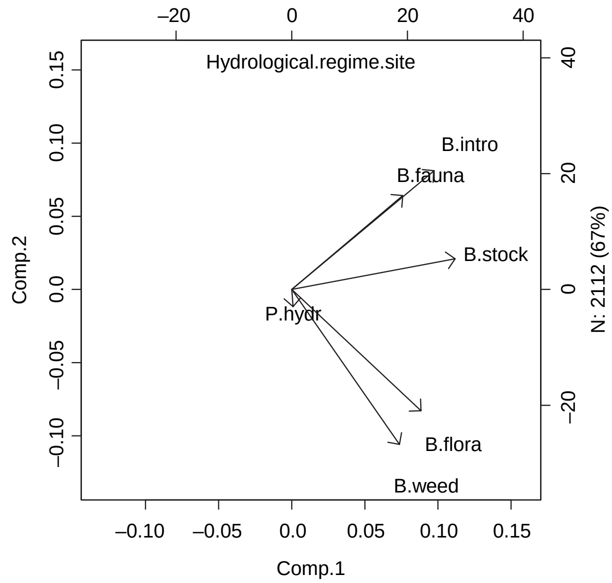

Figure 3. Principal Component Analysis with five biological variables and the variable 'hydrological regime' (P.hydr).

biological variables and the variable C.toxic (Fig. 2), and hydrological regime (Fig. 3) indicate that the non-biological variables were not well represented. Similar results were observed with the other two core variables, with the connectivity variables (Fig. 4), the global pressures variables and the other physical and chemical variables. This indicates that the biological variables span a different dimension than the other impact variables. Principal Component Analysis were also performed with the core set variables and the 19 non-core set variables, one variable at the time (Fig. 4). With the exception of P.con.seg (Fig. 4a) and G.updam (impact of upstream dam, Fig. 4c), good correlations were found between the non-core variables and the core set variables. These results indicate that the core set pressure variables do not account for biolog-

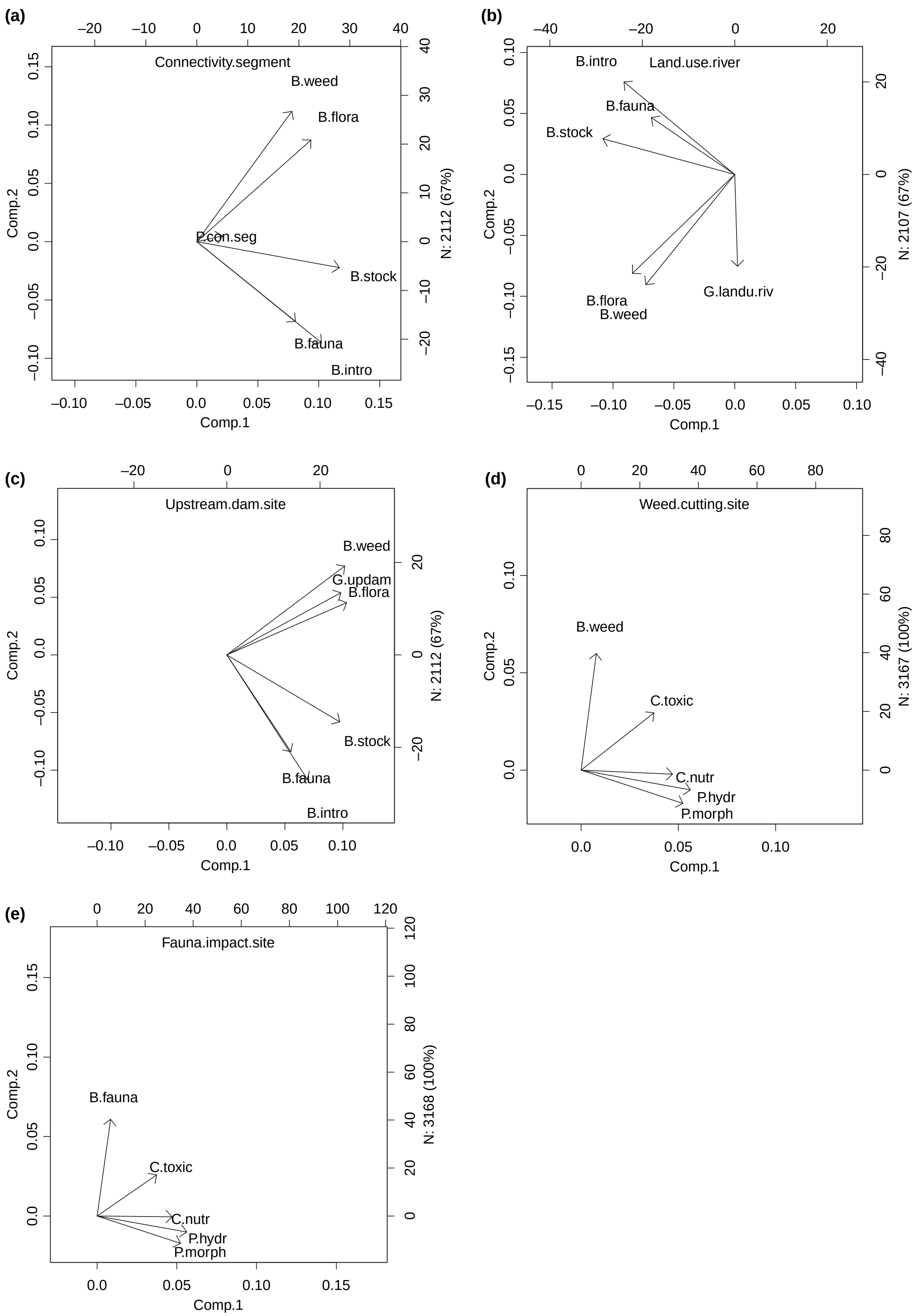

Figure 4. Principal Component Analysis with the four core set variables and (a) connectivity at the segment level (P.con.seg), (b) land use at the river scale (G.landu.riv), (c) upstream dam (G.updam), (d) weed-cutting at the site (B.weed), and (e) impact of other fauna at the site (B.fauna).

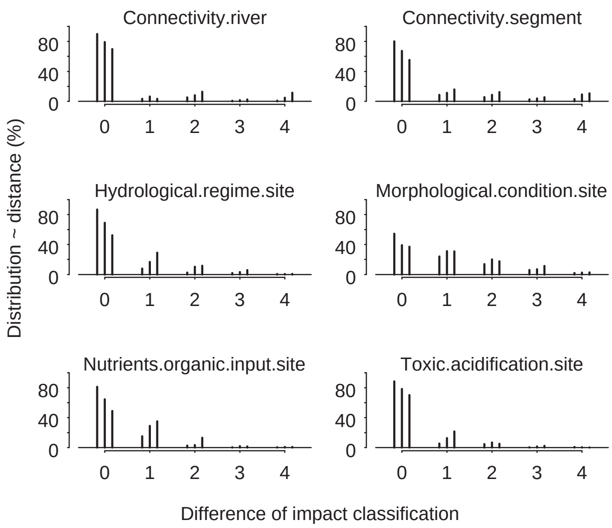

Figure 5. Bar plots showing the global distribution (%) of the variable differences between classification of impact of sites in the same river. Vertical lines give the percentage by three distance classes: 0–3, 3–6 and >6 km.

ical pressures, segment level connectivity and upstream impact of dams.

Correlation between sites within the river

The WFD states that the biological quality elements should reflect the whole catchment, thus only rivers with data from at least two sampling sites were considered. For these rivers average distance between the sampling sites (subdivided into three classes: 0–3, 3–6 and > 6 km) and the maximum difference (ranging from 0 to 4) between each human impact variable were calculated. Small differences were found between the individual impact scores for most variables, except for P.morph (morphological condition; Fig. 5). For this variable, and to some extent also for P.ripz.seg, there was a systematic relation with distance class, with the similarity declining with sites further apart. The variable P.morph differed in almost 60%, and P.ripz. seg in almost 50% of cases between sites within the same river, indicating that these variables were more site-specific than the other variables.

Potential for using low-resolution GIS data

To evaluate if low-resolution GIS data can be used as a simple way to pre-classify sites for impact, two additional GIS variables (land use and population density) were included. As the aim was to retain a large number of sites for spatial analysis, only pressures covering connectivity and physical modifications (P.morph, P.hydr, P.con.seg and P.con.ms), chemical impacts (C.toxic and C.nutr), and the two GIS variables of 1 km land use (GP.landuse.gis) and log of 1-km population (GP.lnpopd.gis) were selected. This gave a data set comprising 6430 sites. Most of these variables were partially correlated with each other, and particularly those measuring similar forms of pressures. The two connectivity variables (P.con.seg and P.con.ms) were highly correlated (r = 0.62), as were the chemical pressures (C.toxic and C.nutr), the two GIS variables (lnGP.popd.gis and GP.landuse.gis) and the physical modification variables (P.morph and P.hydr) (Table 2).

The first two axes of the cPCA explained, respectively, 40% and 16% of the variance in the data. The first axis was positively correlated with all the impact variables. Significant variable contributions were, in decreasing order: C.nutr, lnGP.popd.gis, P.morph, and to a lesser extent C.toxic, P.hydr and P.con.seg (Fig. 6). The second axis was positively correlated with high population densities and land use pressures (urban, crops) and negatively with connectivity modifications. In short, the first axis formed a gradient of degradation and the second axis differentiated between more developed areas (urban or agricultural, with high

Table 2. Pearson's correlation matrix of the relationships between the 8 impact variables of the centred Principal Component Analysis

	P.con.seg	P.con.ms	P.hydr	P.morph	C.toxic	C.nutr	InGP.popd.gis	GP.landugis
P.con.seg	1.000	0.622	0.313	0.312	0.353	0.364	0.338	0.208
P.con.s	0.622	1.000	0.285	0.158	0.279	0.305	0.109	0.012
P.hydr	0.313	0.285	1.000	0.482	0.209	0.308	0.343	0.209
P.morph	0.312	0.158	0.482	1.000	0.240	0.423	0.542	0.334
C.toxic	0.353	0.279	0.209	0.240	1.000	0.595	0.316	0.222
C.nutr	0.364	0.305	0.308	0.423	0.595	1.000	0.473	0.362
InGP.popd.gis	0.338	0.109	0.343	0.542	0.316	0.473	1.000	0.520
GP.landugis	0.208	0.012	0.209	0.334	0.222	0.362	0.520	1.000

Abbreviations according to text and Table 1

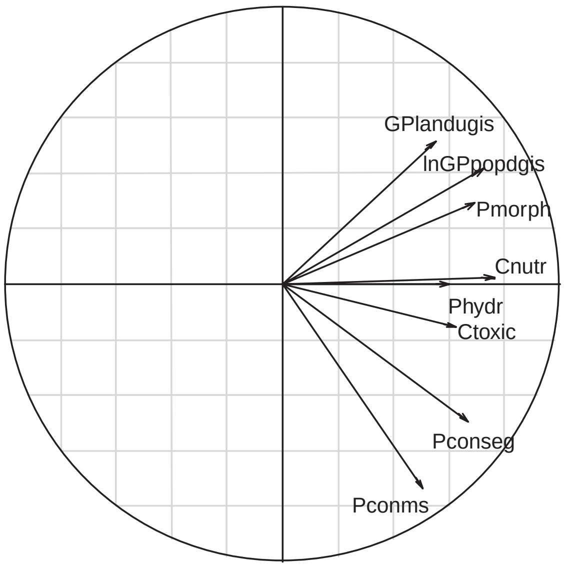

Figure 6. Centred Principal Component Analysis of eight human impact variables (6430 sites) – correlation circle of the first two Principal Component Analysis axises. Abbreviations according to text and Table 1.

population densities) and more rural to semi-natural areas where altered connectivity was more prevalent.

The four pressures that made the greatest contributions to the first axis of the cPCA were those used to calibrate the European Fish Index model (Pont, Hugueny & Rogers 2007), whereas connectivity, which did not produce strong metric responses and was not included in the modelling approach, contributed more to the second axis. Moreover, the GIS variables both contributed significantly to the first axis, and population density in particular was well correlated with both physical and chemical pressures (r = 0.542 with P.morph and 0.473 with C.nutr).

Status of European running waters

The status of European running waters was classified separately for small (catchment size: < 1000 km^2) and large rivers (catchment size: > 1000 km^2). For physical degradation, connectivity at the river basin scale was classified as obstructed (impact class 3–5) in 60% of sites, but with less obstructions in the larger rivers (Table 3). However, in the larger rivers lateral movements of fish onto the floodplain were obstructed to a greater extent than in smaller rivers. Overall, lateral movements of fish were obstructed at 61% of all sites.

Morphological habitat degradation was frequent, as unsatisfactory conditions (impact class 3–5) were noted at 50% of sites, but with generally better conditions in the smaller rivers (Table 3). Unsatisfactory conditions with regard to the hydrological regime were present at 38% of sites, with worse conditions in larger rivers. Riparian vegetation was generally in bad condition in larger rivers.

Chemical impact in the form of toxic or acidic compounds affecting fish communities was infrequent in larger rivers, but unsatisfactory in 27% of smaller rivers (Table 4). Loading of nutrients or organic matter was high (impact class 3–5) in 49% of all sites.

Biological impact was reported for only a small number of rivers, but because these were distributed widely throughout Europe, some conclusions could be drawn. No introduction or no naturalisation of introduced species was found in 92.5% of sites and reproduction of the latter was only found in 4.5% of sites (Table 5). Although stocking of fish was not very common, it was thought to have impact on the fish community to levels beyond good status at 7% of sites.

Table 3. Proportion (%) of sites classified as high-good status (impact 1–2), of moderate status (impact 3) and poor-bad status (impact 4–5) with regard to physical pressure and impact variables. Rivers divided into small (< 1000 km^2 catchments) and large (> 1000 km^2). Abbreviations according to Table 1

		Impact classification			
Variable	Catchment size	1–2	3	4–5	*n*
Connectivity river	Small	34.5	18.2	47.3	5357
(P.con.riv)	Large	63.8	17.6	18.6	1259
Lateral movements	Small	44.5	11.8	43.7	2947
(P.lmov.seg)	Large	25.5	32	42.5	1019
Hydrological regime	Small	67.9	17.1	15	5256
(P.hydr)	Large	38	38.3	23.7	1258
Riparian zone	Small	36.7	18.5	44.8	3758
(P.ripz.seg)	Large	18.4	24.1	57.5	1085
Morphological	Small	55.1	24	20.9	5708
(P.morph)	Large	28.4	14.4	57.2	1325
Sediment load	Small	86.3	3.9	9.8	3688
(P.sload.seg)	Large	92	6.9	1.1	1050

Table 4. Proportion (%) of sites classified as high-good status (impact 1–2), of moderate status (impact 3) and poor-bad status (impact 4–5) with regard to two chemical impact variables at the site scale

		Impact classification			
Variable	Catchment size	1–2	3	4–5	*n*
Toxic or acid	Small	73.4	13.5	13.1	5789
(C.toxic)	Large	91.5	5.9	2.6	1326
Nutrients and	Small	54.6	19.6	25.8	5586
organic (C.nutr)	Large	34.8	59.8	5.4	1320

Table 5. Proportion (%) of sites classified as high-good status (impact 1–2), of moderate status (impact 3) and poor-bad status (impact 4–5) with regard to three biological impact variables at the site scale. Abbreviations are according to Table 1

Variable	Catchment size	Impact classification			
		1–2	3	4–5	*n*
Introduction of fish (B.intro)	Small	89.5	4	6.5	1356
	Large	97.8	0.9	1.3	784
Stocking of fish (B.stock)	Small	92.6	3.3	4.1	1349
	Large	93.7	5.7	0.6	790
Exploitation of fish (B.expl)	Small	84.4	9	6.6	1238
	Large	54.3	36.7	9	267

Exploitation of fish was considered to affect fish fauna of larger rivers more than small rivers (Table 5).

In general, mountainous or upland areas (Alps, Pyrenees, French Massif Central, Ardennes, western UK, Scandinavian mountain range) were characterised by lower pressure scores, whereas lowland areas were characterised by comparatively higher impacts. The most heavily impacted areas appear to correspond to the most heavily populated and developed areas in Europe.

Discussion

The basis for classification of human impact is seldom reported, probably as this process inevitably involves expert judgement, which might be conceived as subjective, although it is considered an important tool in degradation classification (Haunia 2002). Pressures that were mainly classified from expert judgement were well correlated with pressures classified from physical or chemical measurement, or even low-resolution GIS data. This indicates that a classification process employing defined variables and fixed class boundaries using expert judgement is a valid operational method, even on a continental scale.

Notwithstanding, more objective data are also required to support the expert judgement. At the river basin and sub-basin level, GIS data on land use are important indicators of pressure (Wang *et al.* 2000). The additional two GIS variables (land use and population density) gave good results, despite the low precision of the data and the way the information was integrated (ambient value at segment level). Compared with expert data, these GIS data covers have the advantage of greater uniformity or homogeneity in the calculation and the precision of the estimations. This implies that there would be less bias in standardised GIS data across Europe than expert evaluations on a country or regional level. Given the relatively simple manner in which these low-resolution data were integrated, and the widespread availability of GIS environmental data today, these results are particularly promising.

Global, physical and chemical pressures were generally well correlated, which indicates that the joint impact variable constructed from four pressure variables (core set variables) was a good descriptor of these pressures. Also, expression of these pressures at different scales was well correlated, indicating that for most pressures scale is not of crucial importance. Water quality, hydrological regime and land use could therefore be quantified at a scale larger than the site level.

Global pressures were generally well correlated between sites in the same river; the closer the site, the higher the similarity. Nevertheless, morphological alterations and the status of the riparian vegetation differed considerably between sites, i.e. they were site-specific and not part of the river continuum as were hydrological and chemical properties. Because the assessment of ecological status is based on fish over a large scale most variables should be evaluated on a larger scale. It is known that the morphology and the status of the riparian zone of a chosen site are not always representative for the whole river or even segment, as many sites are chosen based on their accessibility. Often these places are close to a road or next to a bridge, where the morphology and riparian zone can be more impacted than the rest of the segment. It is essential to develop models, preferably GIS-related, for an objective classification of these variables on the larger scales.

The biological pressures were excluded from the calculation of the joint pressure variable because these pressures were not well correlated with physical and chemical pressures or population density. This means that one dimension of human stress on the ecosystem was not properly covered. The main reason for not including biological pressures was lack of information in several countries. It would seem that it is of national interest to have complete registers of introductions and stocking of fish and other biota. One way of circumventing this problem is to gather historical data, which are often present for important fish species and compare the present community with the historical record.

Connectivity was also left out of the joint pressure variable because of its low correlation with fish data and lack of information regarding connectivity for many sites. Again, gathering of historical distribution of long-distance migrators may resolve this problem.

Finally, it appears from the analysis that degradation of European streams is widespread. More than 49% of sites were in moderate to poor condition, mainly as a result of disruption to connectivity (both longitudinal and lateral) and physical habitat condition and increased nutrient and organic loading.

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References

Alabaster J.S. & Lloyd R. (1982) *Water Quality Criteria for Freshwater Fish*, 2nd edn. London: Butterworths, 384 pp.

AQEM Consortium (2002) *Manual for the Application of the AQEM System*. Available at: http://www.aqem.de. [Last accessed 10 October 2007.]

Barbieri R., Economou A.N., Stoumboudi M.Th. & Economidis P.S. (2002) Freshwater fishes of Peloponnese (Greece): distribution, ecology and threats. In: M.J. Collares-Pereira, I.G. Cowx & M. Coelho (eds) *Conservation of Freshwater Fishes: Options for the Future*. Oxford: Blackwell Publishing, pp. 55–64.

Cowx I.G. (2002) Analysis of threats to freshwater fish conservation: past and present challenges. In: M.J. Collares-Pereira, I.G. Cowx & M. Coelho (eds) *Conservation of Freshwater Fishes: Options for the Future*. Oxford: Blackwell Publishing, pp. 201–220.

Degerman E. & Lingdell P.-E. (1993) *pHisces – fish as Indicators of low pH*. Report from the Institute of Freshwater Research No. 3 . pp . 37–54 (in Swedish).

Freyhof J. (2002) Freshwater fish diversity in Germany, threats and species extinction. In: M.J. Collares-Pereira, I.G. Cowx & M. Coelho (eds) *Conservation of Freshwater Fishes: Options for the Future*. Oxford: Blackwell Publishing, pp. 3–22.

Haunia S. (2002) Good ecological status: reference conditions for surface waters. European network of Freshwater Research Organisations, Proceedings. Helsinki, Finland: Euraqua European, Network of Freshwater Research Organisations, 186 pp.

Impress Group (2003) Guidance for the Analysis of Pressures and Impacts. Guidance Document No. 3. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Brussels: Office for Official Publications of the European Communities. 153 pp.

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

Landscan (2003) Available at: http://www.ornl.gov/sci/gist/landscan/landscan2003/index.html, © 2002, UT BATTELLE, LLC.

Paller M.H. (2001) Comparison of fish and macroinvertebrate bioassessments from South Carolina coastal plain streams. *Aquatic Ecosystem Health and Management* **4,** 175–186.

Pont D., Hugueny B. & Rogers C. (2007) Development of a fish-based index for the assessment of river health in Europe: the European Fish Index. *Fisheries Management and Ecology* **14,** 427–439.

Wang L., Lyons J., Kanehl P., Bannerman R. & Emmons E. (2000) Watershed urbanization and changes in fish communities in southeastern Wisconsin streams. *Journal of the American Water Resources Association* **36,** 1173–1175.

Whittier T.R. & Hughes R.M. (1998) Evaluation of fish species tolerance to environmental stressors in lakes in the northeastern United States. *North American Journal of Fisheries Management* **18,** 236–252.

Detecting patterns and relationships of human pressures in European Rivers

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ABSTRACT

Most European rivers are affected by different types of human pressures that may impair fish populations. We analysed 15 pressure variables of 4 different pressure groups, i.e. hydrology, morphology, water quality and connectivity to detect spatial patterns, relationships and interactions between pressures and natural environment at the European scale. Based on literature, national databases and expert knowledge important pressures were identified and collected within the EU-project EFI+ in 14 countries at about 10 000 fish-sampling sites in Europe. In 90% of the catchments analysed fish migration was interrupted by barriers. We used PCA and correlation analysis to identify key pressures and to eliminate redundant pressures at local and river segment scale. Thirteen variables were found to describe the majority of human degradation at a specific site. To aggregate into pressure type specific indices we first harmonized the variables along a gradient from 1-5, i.e. from nearly undisturbed to strongly impacted sites. Further, we calculated the mean of values > class 2 only, to avoid that values <=2 compensate values >2, i.e. to better indicate degradation. Pressure analysis showed that 24% of sites are affected by single, 22% by double 19% by triple and 12% by four pressure groups. Only 23% of sites are less affected, i.e. class <=2. In terms of pressure types, analysed sites showed alterations in 55% for water quality pressures, 40% for hydrology, 37% for morphology and 34% for connectivity (river segment). In 45% of the cases water quality problems are also associated with other pressures. The results clearly show that European rivers are multi-impacted. Therefore, only restoration strategies simultaneously considering all important types of pressures will guarantee the achievement of the good ecological status or potential sensu EU Water Framework Directive.

Key words: multi-impacted rivers, European scale, ecological status, pressure index

1 INTRODUCTION

A number of human alterations - herein after referred to as pressures - directly affecting the physico-chemical conditions of running waters, have a strong influence on fish communities. In European rivers, the most important pressure signifi-

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cantly affecting fish is water pollution (FAME 2004, Degerman et al. 2007). Hydrological alterations as impoundment (Reid 2004), water abstraction (Pyrce 2004) and hydropeaking (Flodmark et al. 2004) are known to degrade fish communities. Morphological alterations such as channelisation (Aarts et al. 2004) and river bed degradation (Raat 2001) also have deleterious effects. Finally, disruption of both longitudinal (Rieman & Dunham 2000) and lateral (Hughes & Rood 2003) connectivity significantly impairs fish communities.

Recently, studies in Europe and worldwide emphasize the influence of different human pressures on rivers and it clearly has been demonstrated that a better understanding of the distinct effects of single pressures, multiple pressures and their interactions is a pre-condition for effective river restoration (Schmutz et al. 2007). But due to the traditional focus on single case studies, basins, watersheds or ecoregions, there is a lack of common understanding of pressures across Europe, though the European Water Framework Directive (WFD, EU 2000) requires a consistent and comparable "identification of significant anthropogenic pressures and the assessment of their impacts on water bodies" (ANNEX II, WFD).

In the IES report (European Commission 2006, Institute for Environment and Sustainability) it's indicated that pressures act simultaneously in most cases and that managers must define a hierarchy amongst these to identify priority actions. The few existing studies, examining relationships between pressures are suggesting strong influences and interactions between two or more kinds of pressures. According to Vinebrooke et al. 2004, pressures rather often have comparative, additive and multiplicative effects. Despite this, only a few studies have focused on the importance of that topic, especially in context of the WFD, dealing with pressure combinations, large datasets or multiple pressures and taking interactions of pressures into consideration.

In this paper we analyse different types of pressures from 15 European countries and 16 ecoregions. Our primary objectives are (1) to identify various pressure groups, (2) to detect dominating pressures (chemical–physical pressures vs. hydromorphological pressures), (3) to analyse multiple pressures and prevailing pressure combinations and (4) to detect spatial patterns of pressures across Europe.

2 METHODS

Dataset

A database prepared and maintained by the consortium of the EU project EFI+ (**http://efi-plus.boku.ac.at/**) was used to quantify human pressures for our study. The EFI+ database is a pan-European database and contains data on fish assemblages, environmental characteristics and human pressures in 15 European countries.

Related to the compulsive "Characterisation of river basins", (Article 5, WFD), a lot of different pressure information has been gathered by EU-member countries since 2004, which was used for this pressure analysis. In addition, regional and national monitoring programmes and profound protocol data from field mappings were available. Qualified pressures for our analysis must be scientifically proven, must have potentially large negative effects on fish and must be relevant on a continental scale. We are aware that land use also has strong indirect influences on rivers, but due to our focus on direct effects of pressures on the river, we only considered instream variables for pressure analysis. In total, 15 pressure variables out of the EFI+ database were qualified for our analyses.

Pressure variables

In total, pressure data at 10208 sites in about 4800 rivers and 16 ecoregions (Table 1) were available for our study. As more than 90% of sites showed continuum disruption at the catchment level, this variable was not used for further analyses.

Table 1: Number of analysed sites per country and ecoregion (NoData represents sites where ecoregion classification was not available/possible).

		Country abbreviation															
		AT	CH	DE	ES	FI	FR	HU	IT	LT	NL	PL	PT	RO	SE	UK	Total
Ecoregions	Alps	371	163				52		88								**674**
	Borealic uplands					12									61		**73**
	Central highlands	439	2	289			14					23					**767**
	Central plains			440							76	414			326		**1256**
	Eastern plains											176		72			**248**
	England															1228	**1228**
	Fenno-scandian shield					251									207		**458**
	Hungarian lowlands	63						163									**226**
	Ibero-Macaronesian region				2077								922				**2999**
	Italy and Corsica		9				5		338								**352**
	NoData		106	30		15	66	28	72	94	38	235	1	15	11		**711**
	Pontic province													19			**19**
	Pyrenees				20		18										**38**
	The Carpathiens							2				59		157			**218**
	Western highlands		220				219										**439**
	Western plains			22	1		411				68						**502**
	Total	**873**	**500**	**781**	**2098**	**278**	**785**	**193**	**498**	**94**	**182**	**907**	**923**	**263**	**605**	**1228**	**10208**

First, we summarized the selected pressure variables in 4 groups, i.e. connectivity (2 variables), hydrology (5), morphology (5) and water quality (3). Information on pressure intensity was available in verbal ordinal form, ranging from 2 to 5 modalities (Table 2). To overcome this inequity, we defined an ordinal ranking scheme and harmonized all pressure variables along a gradient from 1-5, i.e. from nearly undisturbed to strongly impacted sites. In the next step, Principle Component Analysis (PCA) was used to identify key and redundant pressures. PCA was done for all pressure variables in one step and then for each pressure group. Finally, 13 variables remained for further analyses (see Table 2).

Table 2: Human pressure variables selected for pressure analyses, separated in the groups hydrology (H), morhphology (M), water quality (W) and connectivity (C).

Pressure variable	Group	Code	Explanation; short description of classes
Impoundment	H	H_imp	Natural flow velocity reduction on site due to impoundment; 1 = no (no impoundment), 3 = weak, 5 = strong
Hydropeaking	H	H_hydrop	Site affected by hydropeaking; 1 = no (no hydropeaking), 3 = partial, 3 = yes
Water abstraction	H	H_waterabstr	Site affected by water flow alteration/minimum flow; 1 = no (no water abstraction), 3 = weak to medium (less than half of the mean annual flow), 5 = strong (more than half of mean annual flow)
Reservoir flushing	H	H_resflush	Fish fauna affected by flushing of reservoirs upstream of site; 1 = no, 3 = yes
Hydrograph modification	H	H_hydromod	Seasonal hydrograph modification due to hydrological alteration (water storage for irrigation, hydropower etc.); 1 = no, 3 = yes
Channelisation*	M	M_channel	Alteration of natural morphological channel plan form; 1= no, 3 = intermediate, 5 = straightened
Cross section*	M	M_crosssec	Alteration of cross section; 1 = no, 3 = intermediate, 5 = technical crossec./U-profile
Instream habitat*	M	M_instrhab	Alteration of instream habitat conditions; 1= no, 3 = intermediate, 5 = high
Embankment	M	M_embankm	Artificial embankment; 1 = no (natural shoreline), 2 = slight (local presence of artificial material for embankment), 3 = intermediate (continuous embankment but permeable), 5 = high (continuous, no permeability)
Flood protection	M	M_floodpr	Presence of dykes for flood protection; 1 = no, 3 = yes
Barriers segment upstream	C	C_B_s_up	Barriers on segment level upstream; 1 = no, 3 = partial, 3 = yes
Barriers segment downstream	C	C_B_s_do	Barriers on segment level downstream; 1 = no, 4 = partial, 4 = yes
Acidification	W	W_acid	Acifidication; 1 = no, 3 = yes
Eutrophication	W	W_eutroph	Artificial eutrophication; 1 = no, 3 = low, 4 = intermediate (occurrence of green algae), 5 = extreme (oxygen depletion)
Organic pollution	W	W_opoll	Is organic pollution observed; 1 = no, 3 = weak, 5 = strong
* Variables summed up into M_morhph_instr (Mean of M_channel, M_instrhab, and M_crossec)			

To evaluate the status of European rivers in terms of pressure type, 4 pressure type specific indices (hydrological, morphological, water quality and connectivity) were aggregated. They were calculated by averaging values > class 2 only, to avoid that values <=2 compensate values >2, i.e. to better indicate degradation.

Combination of pressures

To focus on the degradation of European rivers related to different types of single/ multiple pressures, we analysed typical combinations of pressures across Europe:

Water quality pressures only, hydromorphological pressures only and a combination of the two types (W, HMC, W + HMC). For this analysis, the pressure type specific indices have been used.

3 RESULTS

Pressure type specific indices

Water quality pressures were detected in 55.9 % of sites (Figure 1), with worst conditions for sites in the Netherlands and Germany (90% of sites affected by water quality pressures). For hydrological pressures, impacts were classified in 40% of sites with worst conditions for sites in the Netherlands, Germany, France, Italy, Portugal and UK (about 50% of sites are impacted by hydrological pressures).

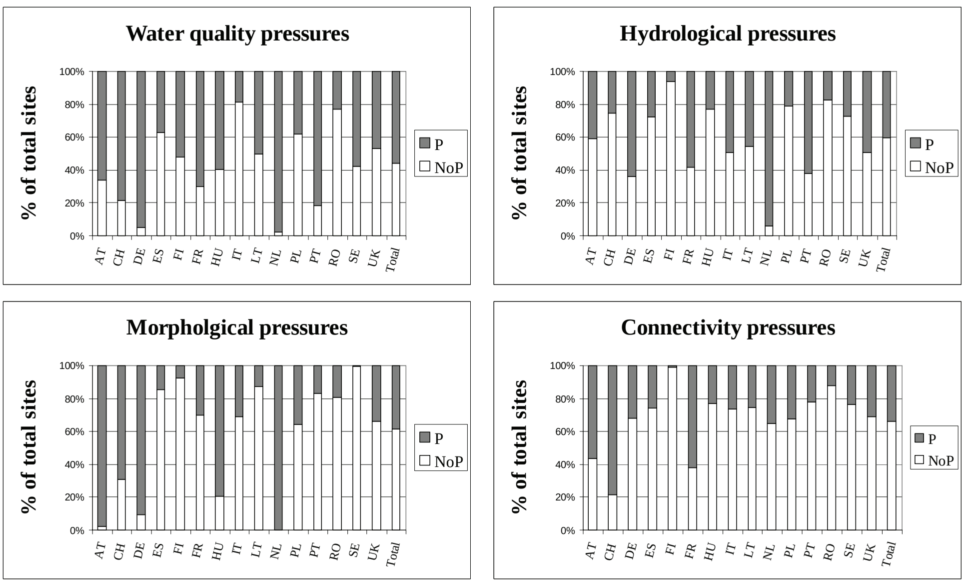

Figure 1: Results of pressure type specific indices in % of sites per country, P = impacted by pressure, NoP = no pressure (class <=2).

Morphological habitat degradation was frequent, as unsatisfactory conditions were noted at 38% of sites with worst conditions for sites in the Netherlands, Austria, Germany, Switzerland and Hungary (more than 50% of sites are related to impact class 3-5 for morphological pressures). Connectivity pressures were reported for only 34% of sites with worst conditions for sites in Austria, Switzerland and France (more than 50% of sites are impacted by connectivity pressures, Figure 1).

Combination of pressures

In 45% of the cases water quality problems are also associated with other pressures and only 11 % of sites are affected by water quality problems only (Figure 2).

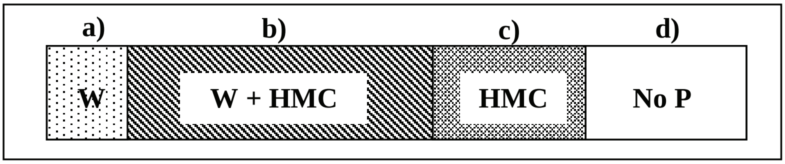

Figure 2: Proportion of sites affected by (a) water quality pressures only (11 %), (b) by water quality and hydromorphological pressures (about 45%), (c) by hydromorphological pressures only (21%) and (d) nearly undisturbed sites (about 23%).

Combined pressure analysis showed that patterns and relationships vary throughout Europe. Combined pressures (W+HMC) are frequent at sites in Austria, Switzerland, Germany, France, Netherlands and Portugal (more than 50% of sites). Hydromorphological pressures without significant water quality pressures can be detected at 47% of Italian sites, only water quality pressures at about 40% of Swedish sites. All other countries do not show specific patterns of pressure combinations (Figure 3).

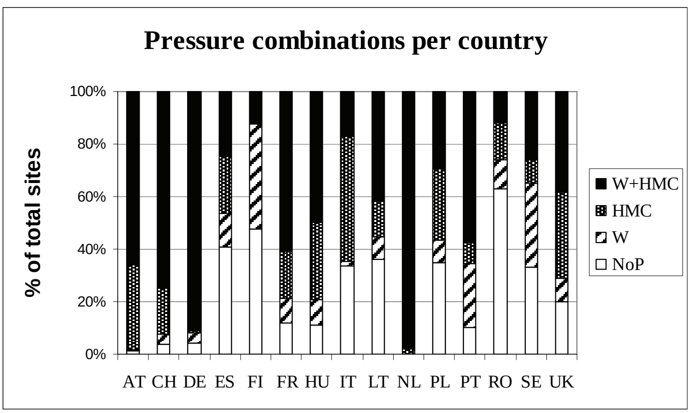

Figure 3: Pressure combinations per country

For the analysis of pressure combinations in different European ecoregions, the output was that particular the Alps are affected by hydrological impacts in combination with morphological degradation. In the Central highlands, almost all sites are affected by hydro-morphological as well as water quality pressures.

Combined pressure analysis also showed that 24% of sites are affected by only one, 22% by two, 19% by three and 12% by four pressure groups. Only 23% of sites are not affected, i.e. class <=2. Especially German, Swiss and Dutch sites are affected by three or four pressure groups (around 40% of sites and more).

4 DISCUSSION

Our intent in analysing pressure variables was to classify river sites by pressure type specific indices, able to separate highly disturbed sites from slightly disturbed sites. This exercise worked well and the pressure type specific indices were able to detect high frequent combination types as hydro-morphological pressures.

Clear limitations in our study were differences in data quality between countries and data sources. Especially the level of detail and the categorization of pressures vary among countries. We tried to overcome this problem by harmonising pressure information into an ordinal ranking scheme, but nevertheless some uncertainty remains. Another problem is that data are not always representative for all countries and ecoregions because of their spatial unequal distribution (e.g. Romania and Spain, Figure 1).

Finally, it appears from our analysis that degradation of European streams is widespread. More than 76% of sites were in moderate to poor pressure condition, mainly as a result of water quality pressures in combination with other pressures.

5 CONCLUSIONS

This study has demonstrated that the different indices allow comparison of pressure status across a large spatial range of countries and river types. Further, the pressure type specific indices have shown that they can distinguish between impacted and unimpacted sites. However, further efforts must be put in the accuracy of pressure data and the compilation of common databases. In-depth examination of relationships among different types of pressures and the linkages to biotic classifications may help reveal a better understanding of restoration and mitigation requirements.

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References

Aarts, B. G. W., Van den Brink F. W. B., and Nienhuis P. H., (2004). „Habitat loss as the main cause of the slow recovery of fish faunas of regulated large rivers in Europe: the transversal floodplain gradient”. *River Research and Applications* 20, 3-23.

Degerman, E., Beier, U., Breine, J., Melcher, A., Quataert, P., Rogers, C., Roset, N., and Simoens, I., (2007). "Classification and assessment of degradation in European running waters". *Fisheries Management and Ecology* 14, 417-426.

European Commission (2006), "Indicators and methods for the ecological status assessment under the Water Framework Directive", EUR 22314EN – DG Joint Research Centre, Institute for Environment and Sustainability, Editors: Angelo G.

search Centre, Institute for Environment and Sustainability, Editors: Angelo G. Slimini, Ana Cristina Cardoso and Anna-Stina Heiskanan, Luxembourg: Office for Official Publications of the European Communities, 2006 – VIII, 252pp. – 17 x 24 cm, Scientific and Technical Research series, ISBN 92-79-02646-1.

EU (2000). "European Union - Directive 2000/60/EC of the European Parliament and of the Council". *Official Journal of the European Communities.*

FAME-Consortium (2004). "Development, Evaluation & Implementation of a Standardised Fish-based Assessment Method for the Ecological Status of European Rivers - A Contribution to the Water Framework Directive (FAME)". Final Report. EVK1 -CT-2001-00094. http://fame.boku.ac.at/downloads/EVK1-CT-2001-00094_FAME_FinalReport_Section5_6.pdf

Flodmark, L.E.W., (2004). "Hydropeaking - a potential threat or just a nuisance? Experiments with daily discharge fluctuations and their effects on juvenile salmonids". Faculty of Mathematics and Natural Sciences. Oslo, University of Oslo. *A collection of IV papers.*

Hughes, F.M.R., and Rood S.B., (2003). "Allocation of river flows for restoration of floodplain forest ecosystems: a review of approaches and their applicability in Europe". *Environmental Management* 31, 12-33.

Pyrce, R., (2004). "Hydrological low flow indices and their use". Peterborough, Ontario. Watershed Science Center. 33.

Raat, A.J.P., (2001). "Ecological rehabilitation of the Dutch part of the River Rhine with special attention to the fish". *Regulated Rivers-Research & Management* 17, 131-144.

Reid, S.M., (2004). "Post-Impoundment Changes to the Speed River Fish Assemblage". *Canadian Water Resources Journal* 29, 183-194.

Rieman, B. E., and Dunham J.B., (2000). "Metapopulations and salmonids: a synthesis of life history patterns and empirical observations". *Ecology Freshwater Fish* 9, 51-64.

Schmutz, S., Melcher A., Muhar S., Zitek A., Poppe M., Trautwein C., and Jungwirth M., (2007): "MIRR - Model-based Instrument for River Restoration. Entwicklung eines strategischen Instruments zur integrativen Bewertung ökologischer Restaurationsmaßnahmen an Fließgewässern". *Schriftenreihe des Deutschen Rates für Landschaftspflege* 81, 119-122.

Vinebrooke, R. D., and Cottingham K. L. et al. (2004). "Impacts of multiple stressors on biodiversity and ecosystem functioning: the role of species co-tolerance." *Oikos* 104(3): 451-457.

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ORIGINAL PAPER

Assessing the impact of a downscaled climate change simulation on the fish fauna in an Inner-Alpine River

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Abstract This study assesses the impact of a changing climate on fish fauna by comparing the past mean state of fish assemblage to a possible future mean state. It is based on (1) local scale observations along an Inner-Alpine river called Mur, (2) an IPCC emission scenario (IS92a), implemented by atmosphere-ocean global circulation model (AOGCM) ECHAM4/OPYC3, and (3) a model-chain that links climate research to hydrobiology. The Mur River is still in a near-natural condition and water temperature in summer is the most important aquatic ecological constraint for fish distribution. The methodological strategy is (1) to use downscaled air temperature and precipitation scenarios for the first half of the twenty-first century, (2) to establish a model that simulates water temperature by means of air temperature and flow rate in order to generate water temperature scenarios, and (3) to evaluate the impact on fish communities using an ecological model that is driven by water temperature. This methodology links the response of fish fauna to an IPCC emission scenario and is to our knowledge an unprecedented approach. The downscaled IS92a scenarios show increased mean air temperatures during the whole year and increased precipitation totals during summer, but reduced totals for the rest of the annual cycle. These changes result in scenarios of increased water temperatures, an altered annual cycle of flow rate, and, in turn, a 70 m displacement in elevation of fish communities towards the river's head. This would enhance stress on species that rely on low water temperatures and coerce cyprinid species into advancing against retreating salmonids. Hyporhithral river sectors would turn into epipotamal sectors. Grayling (*Thymallus thymallus*) and Danube salmon (*Hucho hucho*), presently characteristic for the Mur River, would be superceded by other species. Native brown trout (*Salmo trutta*), already now under pressure of competition, may be at risk of losing its habitat in favour of invaders like the exotic rainbow trout (*Oncorhynchus mykiss*), which are better adapted to higher water temperatures. Projected changes in fish communities suggest an adverse influence on salmonid sport fishing and a loss in its high economic value.

Keywords Climate change · Fish fauna · Scenarios · River · European Alps

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Introduction

The goal of this study is to contribute to the understanding of how climate change may alter the structure of freshwater ecosystems. This is achieved by a chain of empirical models from climate research to hydrobiology. Thereby we implicate an emission-scenario in the assessment of the impact on the fish fauna rather than ad hoc assumptions.

This appears desirable as it allows us to assign the response of fish to a particular emission scenario, describing a specific way of how mankind may evolve.

Fishes occupy the highest trophic levels in riverine ecosystems—integrating over processes at lower trophic levels—making them ideal indicators of the state of the environment. Fishes are homoeothermic, thus processes structuring fish assemblages are closely linked to water temperature. One of the most pervasive zoogeographic patterns is that of ‘cold-water’ versus ‘warm-water’ fish species, with a sharp sorting of species along altitudinal gradients. In mountainous regions, this leads to a clear longitudinal zoning of riverine fish assemblages from the headwaters to the mouths. In Europe, the concept of fish zones, i.e. trout, grayling, barbel and bream zones, is a well established concept used to describe the longitudinal patterns of fish assemblages (e.g. Thienemann 1925; Huet 1949). This pattern is defined mainly by longitudinal water temperature increase, restricting salmonids to headwaters and cyprinids to low reaches. Salmonid species, e.g. brown trout (*Salmo trutta*), are cold-water species showing a limited tolerance against high-water temperatures (Elliott 2000). Therefore, warming of rivers can be assumed to affect these populations by exceeding temperature preferences and tolerance limits (Rahel et al. 1996; O’Brien et al. 2000; Reid et al. 2001; Hari et al. 2006). On the other hand cyprinid species like chub (*Leuciscus cephalus*) depend on minimum temperature thresholds for reproduction. Global warming is expected to shift cold-water species towards the poles and to higher altitudes (Jackson and Mandrak 2002; Rahel 2002). Space-for-time substitution, using current biogeographic limits to project fish distributions is a methodology applied by several authors to analyse scenarios of water temperature increase (e.g. Eaton and Scheller 1996; Rahel 2002). This approach employs empirical models describing the spatial patterns of fish species or assemblages determined by climatic factors in the past projected into the future using different climate scenarios (e.g. Flebbe et al. 2006).

Based on down-scaled transient atmosphere-ocean global circulation model (AOGCM) simulations (Matulla 2005) and empirical models, scenarios for water temperature—consistent with the IPCC IS92a emission scenario—are derived. A direct link between an IPCC emissions scenario and the response of aquatic biocoenoses is thereby established, allowing the assessment of the impact caused by a potential climatic change onto the Mur River’s fish assemblages.

Materials and methods

Datasets

Study region: the Mur River

The Mur River shows a clear zonation from trout over grayling and barbel to the bream zone. It is one of the longest Austrian rivers, with a total length of 444 km and a catchment area of about 10,000 km^2. Approximately 350 km of its course extends into Austria (Fig. 1). Headwaters are located at an altitude of approximately 1,890 m. Even though the Mur River has been affected by several hydropower plants and is partly embanked, it still repre-

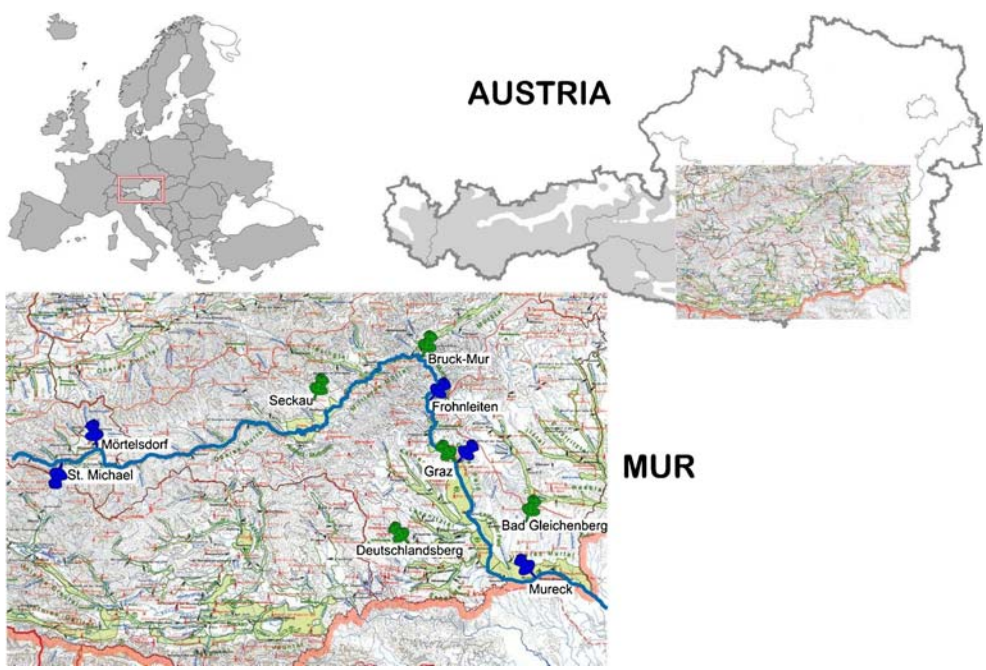

Fig. 1 Austria within Europe and the study region in Austria. The Mur River and stations are highlighted: *blue pins* sites providing water temperature and flow rate, *green pins* air temperature and precipitation recording stations

sents one of the least impacted of the main Austrian rivers (Muhar et al. 1998). Moreover, data on the actual fish fauna (number of species, composition, longitudinal zonation), air temperature, precipitation, water temperature and flow rate are available, which makes the Mur River a suitable candidate for this study.

Observation period (1976–1998)

Observed water temperature and flow rate

Water temperature and flow rate are measured at a riverine group of stations operated by the Austrian hydrographical service (HZB-stations; see Fig. 1, Table 1). Digitized data is available for the period 1976–1998. In addition to the quality testing done by the HZB we carried out further plausibility checks by comparing randomly selected specifically high water temperature anomalies at single sites to measurements at other stations. It was found that they occur together with reasonably high water temperature values at the other stations. The climatological water temperature characteristic throughout the seasonal cycle as derived from the HZB-data is in agreement with typical sub-alpine climatic conditions and thus meets expectations.

For flow rate, we compared specifically high measurements at single stations with measurements at downstream stations and found that they were also pronounced. The HZB-flow rate averages are in accordance with the expected typical characteristics of an Inner Alpine river without glacial influence.

Observed air temperature and precipitation

Monthly mean air temperature and precipitation sums for 1976–1998 are taken from the ALOCLIM dataset (Auer et al. 2001). All station records within ALOCLIM have been subjected to a homogenising procedure (e.g. Auer et al. 2007). The spatial distribution of ALOCLIM-stations used in this study is shown in Fig. 1 and listed in Table 1. The simulation of water temperature at the HZB-stations requires air temperature and flow rate at the HZB-sites. Flow rate is measured on-site but air temperature at the HZB-stations has to be interpolated from the ALOCLIM-stations. Interpolation is done separately for each month (January–December) by linear regression. The Mur River is located within two different climatological regions—the upper reaches in an Alpine valley and the lower reaches are located in flat terrain. For this reason, separate regression models were applied for the upper and lower reaches. Differences between the models are pronounced during the cold season and are probably caused by the inversions that occur in Alpine basins during that time of the year. At lower reaches, 90–97% of the observed temporal variability is explained, while at upper reaches values drop to 77–94%. Haas (2005) presents the approach in detail.

Observed fish distributions

The actual fish fauna of the Mur River consists of more than 40 species, i.e. close to the 40–50 originally occurring species (Table 2), and the observed riverine fish assemblages are close to what could be expected to occur naturally (Schmutz et al. 2004). Currently, 37 species are still reproducing. The lower region, from the Austrian–Slovenian border up to Graz (Fig. 1), belongs to the barbel zone, some low parts even extend into the bream zone. Both rheophilic species including barbel, nase and ide, and stagnophilic species such as rudd, tench, carp or wels are widely spread. The middle reaches, from Graz to Bruck, comprise classical riverine fish species from barbel and grayling zones. Barbel and nase reproduce successfully, as do Danube salmon. Grayling dominates at higher altitudes but also occurs downstream of Graz. Of the 22 historic fish species, 21 still exist in this river section. The river section

Table 1 Downstream stations (in order) with fish zone and the observed fish zone index (FiZI)

Type[a]	Station	lon	Latitude	Altitude (m a.s.l.)	ds[b]	Fish zone	FiZI
HZB	St. Michael	13.64	47.09	1,042	37	Epi-/Metarhithral	3.3
HZB	Mörtelsdorf	13.70	47.13	1,010	51	Metarhithral	3.8
ALOCLIM	Seckau	14.78	47.28	874			
ALOCLIM	Bruck/Mur	15.27	47.42	489			
HZB	Frohnleiten	15.33	47.27	415	137	Hyporhithral/Epipotamal	5.5
HZB	Graz	15.43	47.08	341	347	Hyporhithral/Epipotamal	5.6
ALOCLIM	Deutschlandsberg	15.22	46.83	410			
HZB	Mureck	15.79	46.71	224	408	Epipotamal	6.0
ALOCLIM	Bad Gleichenberg	15.90	46.87	303			

[a] HZB stations are characterized by water temperature [°C], flow rate [$m^3 s^{-1}$], fish zone and calculated FiZI. At ALOCLIM stations, precipitation and air temperature are available for 1976–1998 and the scenario period (2027–2049)
[b] Distance from the source (km)

Table 2 Current occurrence of different fish species and their species-specific fish index ($FI_{sp.}$) from river Mur ($FI_{sp.}$ is calculated for native species only, see also Schmutz et al. 2000)

Species name	Scientific name	Native/exotic	$FI_{sp.}$
Common bream	*Abramis brama*	Native	6.6
Spirlin	*Alburnoides bipunctatus*	Native	6.4
Bleak	*Alburnus alburnus*	Native	6.5
Asp	*Aspius aspius*	Native	6.2
Stone loach	*Barbatula barbatula*	Native	5.5
Barbel	*Barbus barbus*	Native	6
Prussian carp	*Carassius auratus*	Native	6.4
Crucian carp	*Carassius carassius*	Native	6.5
Nase	*Chondrostoma nasus*	Native	5.9
Spined loach	*Cobitis taenia*	Native	6.3
Bullhead	*Cottus gobio*	Native	4
Grass carp	*Ctenopharyngodon idella*	Exotic	
Common carp	*Cyprinus carpio*	Native	6.5
Northern pike	*Esox lucius*	Native	6.2
Ukrainian brook lamprey	*Eudontomyzon mariae*	Native	5.1
Three-spined stickleback	*Gasterosteus aculeatus*	Exotic	
Whitefin gudgeon	*Gobio albipinnatus*	Native	6.5
Gudgeon	*Gobio gobio*	Native	6
Danube salmon	*Hucho hucho*	Native	5.7
European brook lamprey	*Lampetra planeri*	Native	4.5
Pumpkinseed	*Lepomis gibbosus*	Exotic	
European chub	*Leuciscus cephalus*	Native	6
Eurasian dace	*Leuciscus leuciscus*	Native	6.3
Soufie	*Leuciscus souffia*	Native	5.4
Burbot	*Lota lota*	Native	5.6
Rainbow trout	*Oncorhynchus mykiss*	Exotic	
Eurasian perch	*Perca fluviatilis*	Native	6.7
Stone moroko	*Pseudorasbora parva*	Exotic	
Ten-spined stickleback	*Pungitius pungitius*	Exotic	
Bitterling	*Rhodeus sericeus*	Native	6.5
Roach	*Rutilus rutilus*	Native	6.4
Brown trout	*Salmo trutta fario*	Native	3.8
Lake trout	*Salmo trutta lacustris*	Native	3.8
Brook trout	*Salvelinus fontinalis*	Exotic	
Zander	*Sander lucioperca*	Native	6.7
Rudd	*Scardinius erythrophthalmus*	Native	6.7
European grayling	*Thymallus thymallus*	Native	5
Tench	*Tinca tinca*	Native	6.7
Danube streber	*Zingel streber*	Native	6
Zobel	Abramis sapa	Native	6.6
Balkanian barbel	Barbus peloponnesius	Native	6.0
Ide	Leuciscus idus	Native	6.4
White bream	Abramis bjoerkna	Native	6.7
Zingel	*Zingel zingel*	Native	6.3

above Bruck belongs to the trout and grayling zone. Brown trout and grayling are the formative species. The upper limit of Danube salmon distribution is Murau (Schmutz et al. 2000), which is ~50 km downstream of St. Michael (see Fig. 1). Further species are nase, barbel, chub, bullhead, and gudgeon. Today, 13 of the 14 historic fish species inhabit this part of the river. Above Murau the river is dominated by brown trout, which is equivalent to upper and lower trout zone.

Electric fishing data from 1990 to 2000, assembled to a set of 25 measurements (Fig. 3a), were provided by the Institute of Hydrobiology and Aquatic Ecosystem Management, BOKU, Austria. Data were collected along a river section of 332 km in length starting 22 km downstream from the river source. Fish were sampled from a boat with ten electrodes mounted on a boom in front of the boat according to the 'strip-fishing method' (Schmutz et al. 2001). Each habitat type (e.g. gravel bars, steep banks, riffle and runs) was sampled representatively. Fished area per sample ranged from 480 to 35,000 m^2 (7,500 m^2 on average) and the total area fished was about 38 ha. Channel width and slope of sampled sites varied from 11 to 100 m, and 1.4 to 12 m/km, respectively. The observed distribution and the longitudinal arrangement of fish species are assumed to be representative of the total observation period.

Scenario period (2027–2049)

Downscaled scenarios of air temperature and precipitation

Today, AOGCMs are the proper tool to utilise when simulating the climate system's response to an altered chemical composition of the atmosphere. Plausible, but not necessarily likely, future pathways of the atmosphere's composition are formulated as IPCC emission scenarios (IPCC 2001). Their impact on the climatic system is simulated by AOGCMs, which produce output at grids with a horizontal spacing of typically some hundred kilometres. Matulla (2005) used a scenario based on the IPCC IS92a emission scenario, which is implemented by the ECHAM4/OPY3 climate model (Roeckner et al. 1996), to generate scenarios of air temperature and precipitation at the ALOCLIM-stations. The air temperature scenarios are further interpolated onto the HZB-stations (Haas 2005).

Methods

The generation of water temperature scenarios consistent with the IS92a emission scenario is briefly outlined. Based on observed values, we calibrated a model that estimates water temperature from air temperature and flow rate. From Matulla (2005) we have scenarios of air temperature and precipitation that are consistent with IS92a, but no flow rate scenarios are available. Flow rate scenarios were thus generated from the air temperature and precipitation scenarios. The scenarios of air temperature and flow rate are subsequently used to derive water temperature scenarios. Throughout the whole process we employ multiple linear regression (MLR) models. The applied approach to model monthly flow rate may be suboptimal for certain applications but is sufficient to address the needs of this study. The water temperature–fish assemblage model, used to assess the response of the aquatic biocoenosis, takes into account the long-term average of water temperature only. This fact reduces the demands on the models considerably.

Modelling of water temperature

Water temperature and flow rate are measured at HZB-stations and air temperature is interpolated onto the HZB-stations. This allows the calibration of an MLR for every station and every month in the seasonal cycle:

$$Wt_{s,m}(t) = \alpha_{s,m}At_{s,m}(t) + \beta_{s,m}Fr_{s,m}(t) + \gamma_{s,m} + \varepsilon_{s,m}(t) \quad (1)$$

Wt indicates water temperature; At air temperature; and Fr, flow rate. α, β, γ are the regression coefficients and they depend on m (month; $1 \leq m \leq 12$) and s (station). This setup permits the water temperature scenario to depend on the location of the stations. $\varepsilon_{s,m}(t)$ reflects the part of variability that is not captured by the MLR model and is assumed to be normally distributed about mean zero. t stands for the year, which is between 1976 and 1998 for calibration and between 2027 and 2049 for the water temperature scenarios.

Although reproduction of the temporal evolution is not a prerequisite for our application (as mentioned above) it is still an interesting feature of the model and a coarse assessment is shown in Fig. 2. The performance shows a seasonal cycle and is quite reasonable in summer. Calibration utilised the whole datasets.

Estimating flow rate

The modeling of flow rate is a complex task that generally requires more information than precipitation and air temperature. Singh (1995), for instance, provides a comprehensive compilation of hydrological models. Since our focus is on the long-term mean of water temperature rather than on a simulation of temporal evolution, a simple MLR approach, based on precipitation and air temperature as described by Eq. 1 can be applied. As in the case of water temperature, we roughly evaluated the models' performance when simulating the temporal run of flow rate (not shown). We considered the amount of upstream precipitation for each month as well as any possible accumulated precipitation from previous months to account for the accumulation of snow or other storage effects. February–May temperature sums were used as an approximate indicator for the progress of snowmelt. During summer, and at medium and lower reaches, MLR performance appears reasonable. However, the performance at stations near the river's head highlights the necessity of applying hydrological models rather than regression models, if temporal evolution of flow rate is required.

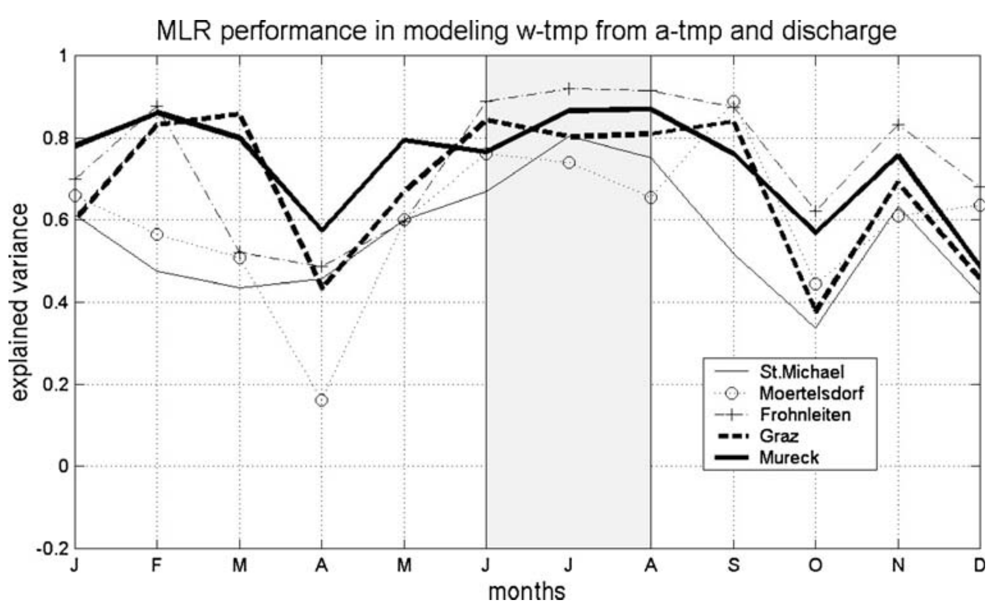

Fig. 2 Evaluation of models' ability to simulate observed water temperature from air temperature and flow rate

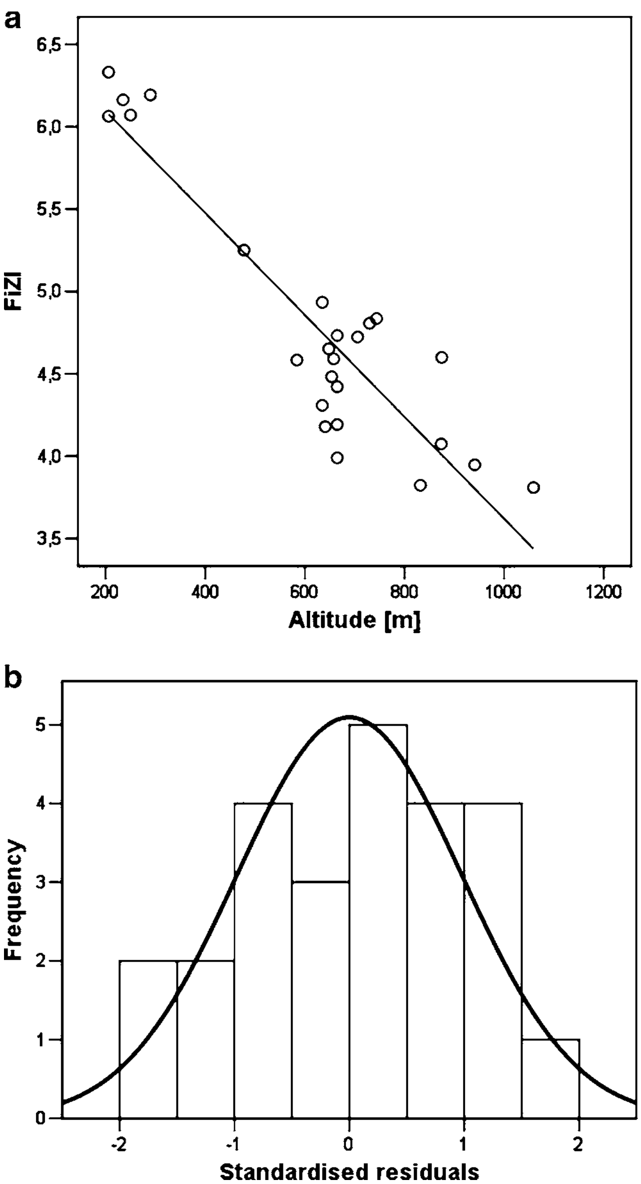

Fig. 3 **a** Scatter diagram of fish zone index (FiZI) versus altitude and the FiZI-altitude regression model used. **b** Distribution of the residuals of the FiZI-altitude regression model

Modeling the fish fauna

Fish zone index The process of associating fish assemblages with the longitudinal structure of running waters often referred to as 'fish zones' has a long tradition (Fritsch 1872). Thienemann (1925) distinguished six fish zones, ranging from the upper brown trout to the ruffe/flunder zone. Since then, similar schemes, such as as biocoenotic regions (Illies and Botosaneanu 1963) and the river continuum concept (Vannote et al. 1980), have been developed (see Table 3). Huet (1949) made the first attempts to describe fish-zones using abiotic descriptors such as slope and river width. Altitude as a surrogate for water temperature has been identified as one of the main environmental parameters structuring fish assemblages along the longitudinal continuum (Flebbe et al. 2006; Pont et al. 2006). Usually, climate change models are developed for a single species, or cold, cool and warm fish assemblages (e.g. Eaton and Scheller 1996; Keleher and Rahel 1996). In order to describe the entire fish assemblage along the river continuum, and for a broader applicability of statistical analyses, the nominal fish-zones were transferred by Schmutz et al. (2000) into a numeric index called Fish Zone Index (FiZI). Table 3 lists all biocoenotic zones from crenal (index value 1) to potamal (index value 7). Regarding Austrian rivers, zones 1 (eucrenal) and 2 (hypocrenal) are most relevant for benthic organisms (Moog and Wimmer 1994), and zones 3 to 7 (epirhithral to metapotamal) for fish.

The species-specific fish index (FI_{sp}) expresses the preference of a species for a fish zone along a river and is calculated as follows:

$$FI_{sp.} = \frac{(3 \times p_3 + 4 \times p_4 + 5 \times p_5 + 6 \times p_6 + 7 \times p_7)}{100}$$

$$\sum_{i=3}^{7} p_i = 100$$

For each fish species, the five fish-zones (indices 3 to 7) are multiplied by their frequency of occurrence. Thus, FI_{sp} ranges from 3 (upper trout zone) to 7 (bream zone). The index of brown trout, for instance, is 3.8, as it prevails in the rhithral, whereas the index of pikeperch is 6.7 as it occurs in the potamal (see Table 3). Each fish zone is defined by its dominating species and typical accompanying species. The FiZI is based on FI_{sp} and describes the composition of fish species at a given sampling site:

$$\mathrm{FiZI} = \frac{\sum_{sp} (N_{sp.} \times FI_{sp.})}{N_{total}}$$

N_{sp} is the number of individuals belonging to a species and N_{total} the number of all caught fish.

FiZI model Water temperature is one of the main factors influencing fish distribution (e.g. Welcomme 1985; Shuter and Post 1990). Highest water temperatures occur

Table 3 Concept of biocoenotic regions, fish-zone and FiZI

Biocoenotic region	Fish-zone	FiZI	Code
Eucrenal	–	1	
Hypocrenal	–	2	
Epirhithral	Upper trout zone	3	ER
Metarhithral	Lower trout zone	4	MR
Hyporhithral	Grayling zone	5	HR
Epipotamal	Barbel zone	6	EP
Metapotamal	Bream zone	7	MP

throughout the summer and hence this period is thought to largely determine the longitudinal distribution of fish species (Le Cren 1955). For this reason, we consider only summer in regards to the impact of climate change on fish fauna. Analyses (not shown) revealed that spatial correlation between water temperatures averaged over 1976–1998 at the HZB stations and the FiZI as derived from electric fishing data is highest during the summer. Averaged water temperature is available at the HZB stations but not for any individual fish sampling site. Therefore altitude is used as a surrogate parameter for long-term mean water temperature and to assign FiZI to the HZB stations. In order to exhaust all available fishing data, an MLR with FiZI as the dependent variable and several abiotic parameters was calibrated. Abiotic parameters known to influence spatial fish assemblage structure, such as altitude [m], distance from source [km], slope [%], size of catchment [km^2], and stream order, are included (Pont et al. 2006). A stepwise linear regression revealed that the model based on altitude alone was the most successful in describing the spatial variability of FiZI ($\alpha<0.001$, R^2=0.82):

$$FiZI(z) = \alpha_1 z + \beta_1 \tag{2}$$

where z stands for altitude. To test Eq. 2, the observations are randomly split into a calibration sample and a validation sample of equal size. α_1 and β_1 are calculated from the calibration period. In the validation period the modeled FiZI is compared to the observed FiZI (Fig. 3b). Modeled and observed FiZI agree and the regression coefficients are highly significant. Hence, Eq. 2 can be used to assign FiZI to the HZB stations (see Table 1). A FiZI between 3.5 and 4.5 from 1,000 m to 800 m a.s.l. indicates the trout zone followed by the transitional zone between trout and grayling (FiZI=4.5) and the grayling zone (FiZI=5.0). The barbel zone (FiZI=6.0) is located around 200 m a.s.l. (Fig. 3a).

To assess the effects of the projected water temperature scenarios (2027–2049) on FiZI, a linear model relating altitude to averaged water temperature (1976–1998) is used. Thereby annual variability is evened out and long-term effects on long-lived species in the Mur River (e.g. Danube salmon) are accounted for. Temporal averaging is indicated by a vertical bar above water temperatures in Eq. 3. To set up the model, FiZI values at the HZB stations calculated via Eq. 2 are inserted into Eq. 3.

$$FiZI(Wt) = \overline{\alpha_2 Wt_z(t)} + \beta_2 \tag{3}$$

Once water temperature projections (2027–2049) are derived, their effect on FiZI is calculated by inserting the corresponding averages into Eq. 3. Based on observed summer water temperatures, the water temperature–FiZI models explain 96% of the spatial variability.

Results

Figure 4a displays the seasonal cycle of observed (1976–1998) and projected (2027–2049) precipitation, averaged over the ALOCLIM-stations. For June and July of each year, projected totals are larger than observed totals, while projected amounts are below observations for the rest of the year. Figure 4b compares and contrasts observed and projected air temperature.

Figure 5a shows the seasonal cycle of flow rate at Mureck. Mureck is the HZB-station farthest downstream, so flow rates at Mureck sum up all upstream effects. Similar to the precipitation scenarios, reductions in the flow rate scenario dominate, with exceptions in March and July. While the latter could be explained by increased precipita-

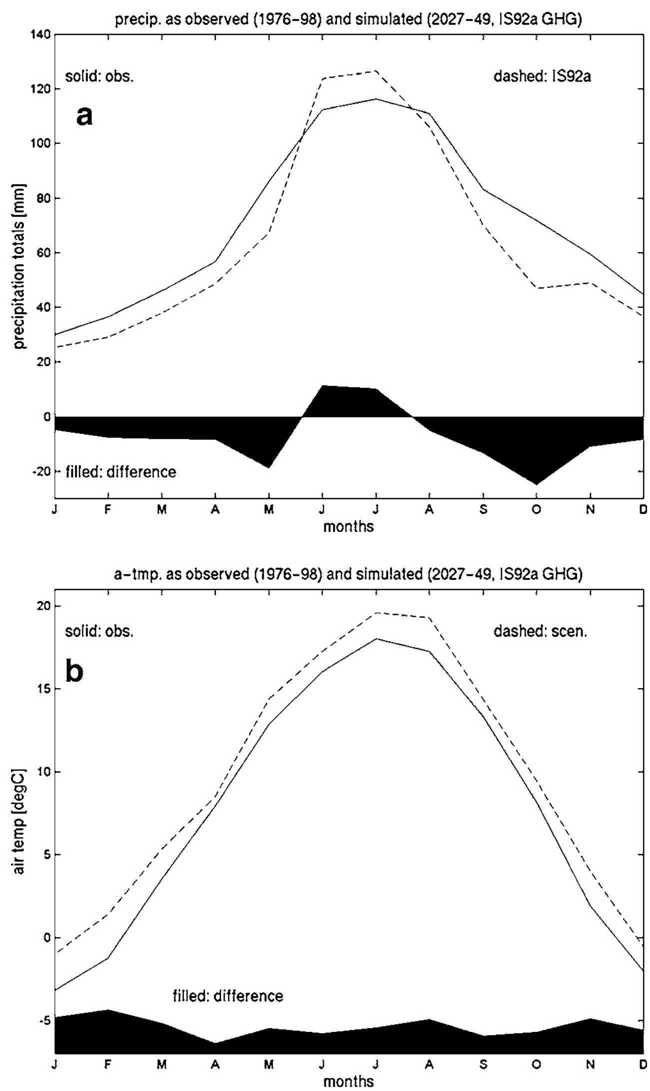

Fig. 4 **a** Precipitation averaged over the ALOCLIM-stations. **b** Air temperature averaged over the HZB-stations. *Solid line* Observations, *dashed line* local scale IS92a GHG scenario, *filled space* scenario minus observations, thus positive values indicate raised values

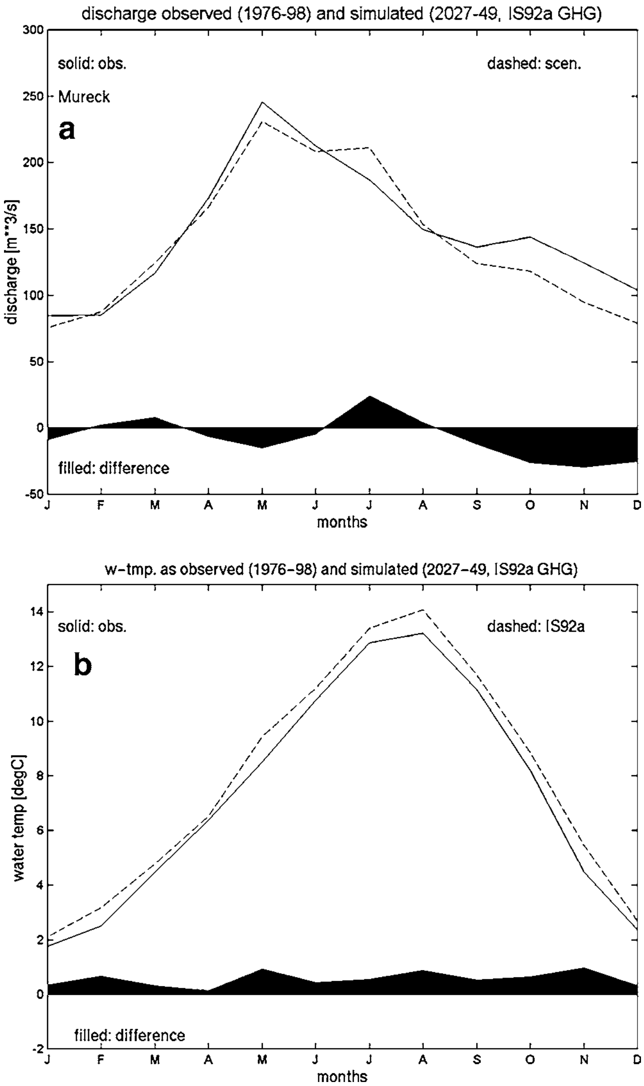

Fig. 5 **a** Flow rate at HZB-station Mureck. **b** Water temperature averaged over the HZB-sites. *Solid line* Observations (1976–1998), *dashed line* local scale IS92a GHG scenario, *filled space* scenario minus observations, thus positive values indicate increasing temperatures

tion sums, the first may indicate an earlier melt off, caused by increased spring temperatures as featured by the scenarios (see Fig. 4b). On the basis of the applied scenarios, the maximum flow rate remains in May. The maximum could have been expected to also advance into April. To examine that possibility in more detail, we analyzed daily flow rate measurements at the HZB-station Graz together with daily air temperature at a station located within the Mur-valley (Zeltweg).

Between 1976 and 1998 the flow rate maximum in Graz occurs at an air temperature of 11.3°C in Zeltweg. If air temperature increases according to the projections shown in Fig. 5a, the flow rate maximum would advance more than 1 week, which is below the monthly time scale referred to here and hence the maximum flow rate remaining in May appears reasonable. However, the point remains that the increased flow rate in mid-summer (as opposed to any other season) has a greater impact on fish fauna because of its influence on water temperature.

Figure 5b compares projected and observed water temperature. As for air temperature, increasing water temperatures are found throughout the whole seasonal cycle. Water temperature features little change in April, which coincides with small changes in air temperature, and pronounced increases in May, August and November. The latter increases can be explained by increased air temperature and reduced flow rate in May, pronounced increase in air temperature in August and November together with no change and pronounced decreases in flow rate, respectively. The greatest impact on fish fauna is the significant ($\alpha<0.05$) increase in water temperature during summer. The amount of warming increases slightly from the metarhithral downstream to the epipotamal/metapotamal zones (not shown). The increase is most evident in August, which features the highest average water temperature. Median values at St. Michael rise from about 9°C to more than 9.7°C, and at Mureck from about 16°C to more than 17°C (not shown).

The averaged water temperature scenarios are entered into Eq. 3. Resulting FiZI values at the HZB sites are significantly different from the observation period ($\alpha<0.05$). The mean increase is 0.22 FiZI units (Fig. 6). To test if changes in FiZI vary with altitude, a regression model between changing FiZI values (projected minus observed) and altitude is set up. The results indicate no trend of FiZI increase along the altitudinal gradient, which appears noteworthy since changes in water temperature along the Mur River increase from headwaters down to lower reaches. The values of FiZI that deviate from the linear

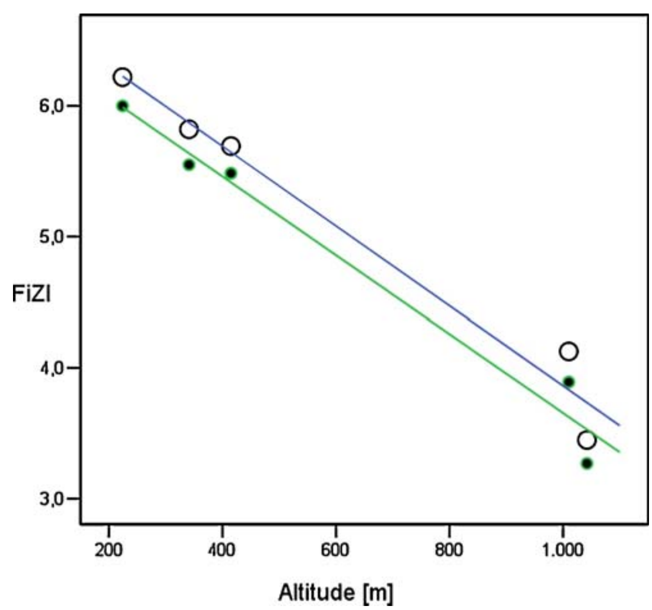

Fig. 6 Evolution of FiZI along the altitude for the observation (*black dots*, 1976–1998) and projection (*circles*, 2027–2049) periods

model within the trout zone (see Fig. 6 at altitudes of about 1,000 m) for both observations and projections may indicate nonlinearities.

Discussion

This study assesses the impact of an IPCC emission scenario on the fish fauna in an Inner Alpine river by comparing the observed fish assemblage to a scenario assemblage that would be reasoned by changes in water temperature according to the IS92a emission scenario. This consistent link from an emission scenario to aquatic biocoenosis is, in our opinion, a novel aspect of this study. The modelled response behaviour of aquatic biocoenosis appears more relevant than if it had been based on ad hoc assumptions regarding water temperature.

The quantification of the change in fish assemblage along the Mur River is regarded as the main result. Focus is placed on summer, when high water temperatures represent the greatest ecological distress for fish fauna (Isaak and Hubert 2004). Findings highlight the importance of water temperature, or altitude as its surrogate, in structuring the distribution of fish species along the river course. This is in line with Pont et al. (2006), who identified altitude as one of the most significant environmental descriptors to discriminate among fish assemblage types and zonation in Europe. The applied water temperature–FiZI model does not require complex modeling of flow rate and water temperature time series as it evaluates the impact on fish fauna using the long-term averaged water temperature.

The temporal averaged water temperature scenarios feature a downstream increasing warming. During summer, the projected warming is about 0.7°C near the riverhead and somewhat more than 1°C at lower reaches. This warming seems compatible with Morrison et al. (2002), who found for the Fraser River (BC, Canada) and a related emission scenario a mean increase in summer water temperature of 1.9°C for the period 2070–2099. The projected change in summer water temperature causes an increase of the FiZI by 0.22 units. This translates into a pronounced elevational shift of about 70 m or, underlying a slope of 2.6 m/km, an upstream transition fish fauna of about 27 km. In Wyoming (USA) a 2°C projected increase in mean July air temperature would result in a 29.1% reduction in the geographic area containing suitable salmonids (Keleher and Rahel 1996). A scenario doubling atmospheric carbon dioxide could cause a 50% loss of cold and cool water fish habitat across the United States (Eaton and Scheller 1996). Unlike the water temperature scenario, the increase in FiZI does not depend on altitude. This indicates that fish species in the trout zone near the headwaters are more temperature sensitive than fish species residing at lower reaches.

According to this scenario, salmonid species are pushed upstream and the barbel and bream zone is elongated. The narrow temperature niche of salmonids makes the significant community shift plausible. The temperature range of brown trout between preferred and lethal conditions is only a few degrees Celsius (Elliott 1994). Therefore brown trout will avoid critical water temperatures or experience higher mortality. Graylings would be greatly affected because habitat factors such as river width or competition with more temperature-tolerant species like the rainbow trout are expected to limit its upward shift (Schmutz 1995). Consequently, this endangered species would face additional threats that could cause its extinction. Most European rivers are fragmented by numerous weirs disrupting the migratory continuum for fish. Therefore the projected upward shift of fish assemblage is not possible, resulting in a general impoverishment of the local fish fauna (Rahel et al. 1996; Hari et al. 2006) unless interventive measures are taken. Salmonid rivers are some of the most economically valuable fishing resources in Austrian waters. A reduction in the extent or quality of salmonid rivers can be expected to have significant effects on the economy of freshwater fisheries.

One field for future research is the description of fish fauna by water temperature. The current use of linear univariate models to describe fish fauna's dependence on water temperature is likely an over simplification enforced by the limited sample of fish data. Enlargement of the sample size by including other rivers could help investigate the relationship between fish fauna and water temperature in greater depth. Moreover, other factors besides water temperature that determine the spatial distribution of fish species should be considered in future studies. This is specifically necessary when translating findings to other rivers. Rahel and Nibbelink (1999), for instance, found that stream size interacts with mean July air temperature and influences the distribution of brown trout in streams located in south-eastern Wyoming (USA). In this latter study, the habitat of the brown trout was associated with areas featuring mean air temperatures of 19–22°C in July. Within this thermal zone, the brown trout was more likely to occur in large streams (>4 m wetted width) than in small streams.

Another issue that should be addressed in future studies deals with the number of scenarios used to assess the impact on fish assemblage and ecosystems in general. Future assessments ought to be based on more than one emission scenario and more AOGCM realisations and downscaling strategies. Results carry more weight when randomisation is applied at every level of uncertainty (e.g. Huitson 1966).

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References

Auer I, Böhm R, Schöner W (2001) Austrian long-term climate 1767–2000-Multiple instrumental climate time series from Central Europe. Central Institute for Meteorology and Geodynamics, Hohe Warte 38, Vienna, Austria

Auer I, Böhm R, Jurkovic A, Lipa W, Orlik A, Potzmann R, Schöner W, Ungersböck M, Matulla C, Briffa K, Jones P, Efthymiadis D, Brunetti M, Nanni T, Maugeri M, Mercalli L, Mestre O, Moisselin J-M, Begert M, Mueller-Westermeier G, Kveton V, Bochnicek O, Stastny P, Lapin M, Szalai S, Szentimrey T, Cegnar T, Dolinar M, Gajic-Capka M, Zaninovic K, Majstorovic Z, Nieplova E (2007) HISTALP-historical instrumental climatological surface time series of the greater Alpine region 1760–2003. Int J Climatol 17:14–46

Eaton JG, Scheller RM (1996) Effects of climate warming on fish thermal habitat in streams of the United States. Limnol Oceanogr 41:1109–1115

Elliott JM (1994) Quantitative ecology and the brown trout. Oxford University Press, Oxford

Elliott JM (2000) Pools as refugia for brown trout during two summer droughts: trout responses to thermal and oxygen stress. J Fish Biol 56:938–948

Flebbe PA, Roghair LD, Bruggink JL (2006) Spatial modeling to project southern Appalachian trout distribution in a warmer climate. Trans Am Fish Soc 135:1371–1382

Fritsch AJ (1872) Die Wirbeltiere Böhmens. Ein Verzeichnis aller bisher in Böhmen beobachteten Säugetiere, Vögel, Amphibien und Fische. Arch Naturwissensch Landesdurchforsch Böhmen 2:1–152

Haas P (2005) Temperatur Extrapolation entlang von Mur und Ybbs. COSREM 21, Internal report, Institute of Meteorology, Vienna, Austria. http://www.boku.ac.at/imp/klima/Literatur/cosrem21.pdf

Hari RE, Livingstone DM, Siber R, Burckhard-Holm P, Güttinger H (2006) Consequences of climatic change for water temperature and brown trout populations in Alpine rivers and streams. Global Chang Biol 12:10–26

Huet M (1949) Apercu des relations entre la pente et les populations piscicoles des eaux courantes. Schweiz Z Hydrol 11:332–351

Huitson A (1966) The analysis of variance. Griffin's statistical monographs and courses, number 18. Griffin, London

Illies J, Botosaneanu L (1963) Problèmes et méthodes de la classification et de la yonation écologique des eaux courantes considerées surtout du point de vue faunistique. Internationale Vereinigung für theoretische und angewandte Limnologie 12:1–57

IPCC (2001) Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press

Isaak DJ, Hubert WA (2004) Nonlinear response of trout abundance to summer stream temperatures across a thermally diverse montane landscape. Trans Am Fish Soc 133:1254–1259

Jackson D, Mandrak N (2002) Changing fish biodiversity: predicting the loss of cyprinid biodiversity due to global climate change. In: McGinn N (ed) Fisheries in a changing climate. American Fisheries Society Symposium 32. American Fisheries Society, Bethesda, MD, pp 89–98

Keleher CJ, Rahel FJ (1996) Thermal limits to salmonid distributions in the rocky mountain region and potential habitat loss due to global warming: a geographic information system (GIS) approach. Trans Am Fish Soc 125:1–13

Le Cren ED (1955) Year to year variation in the year-class strength of *Perca fluviatilis*. Verh Int Verein Limnol 12:187–192

Matulla C (2005) Regional, seasonal and predictor-optimized downscaling to provide groups of local scale scenarios in the complex structured terrain of Austria. Meteorol Z 14:33–47

Moog O, Wimmer W (1994) Comments to the water temperature based assessment of biocoenozic regions according to Illies and Botosaneanu. Verh Int Verein Limnol 25:1667–1673

Morrison J, Quick M, Foreman M (2002) Climate change in the Fraser River watershed: flow and temperature projections. J Hydrol 263:230–244

Muhar S, Kainz M, Schwarz M (1998) Ausweisung flußtypspezifisch erhaltener Fließgewässerabschnitte in Österreich. Bundesministerium für Land- und Forstwirtschaft, Vienna, Austria

O'Brien CM, Fox CJ, Planque B, Casey J (2000) Climate variability and North Sea cod. Nature 404:142

Pont D, Hugueny B, Beier U, Goffaux D, Melcher A, Noble R, Rogers C, Roset N, Schmutz S (2006) Assessing river biotic condition at a continental scale: a European approach using functional metrics and fish assemblages. J Appl Ecol 43:70–80

Rahel F (2002) Using current biogeographic limits to predict fish distributions following climate change. In: McGinn N (ed) Fisheries in a changing climate. American Fisheries Society Symposium 32. American Fisheries Society, Bethesda, MD, pp 99–110

Rahel FJ, Nibbelink NP (1999) Spatial patterns in relations among brown trout (*Salmo trutta*) distribution, summer air temperature, and stream size in Rocky Mountain streams. Can J Fish Aquat Sci 56:43–51

Rahel FJ, Keleher CJ, Anderson JL (1996) Potential habitat loss and population fragmentation for cold water fish in the North Platte River drainage of the Rocky Mountains: response to climate warming. Limnol Oceanogr 41:1116–1123

Reid PC, Borges MD, Svendsen E (2001) A regime shift in the North Sea circa 1988 linked to changes in the North Sea horse mackerel fishery. Fish Res 50:163–171

Roeckner E, Oberhuber J, Bacher A, Christoph M, Kirchner J (1996) ENSO variability and atmospheric response in a global coupled atmosphere-ocean GCM. Climate Dyn 12:737–745

Schmutz S (1995) Zonierung und Bestandsprognose von Bachforelle, (*Salmo trutta* f. fario, L.), Regenbogenforelle, (*Oncorhynchus mykiss*, WAL.) und Aesche, (*Thymallus thymallus*, L.) anhand von Makrohabitatparametern in oesterreichischen Rhithralgewaessern. Dissertation HFA-BOKU, University of Agricultural Sciences, Vienna

Schmutz S, Kaufmann M, Vogel B, Jungwirth M (2000) Grundlagen zur Bewertung der fischökologischen Funktionsfähigkeit von Fliessgewässern. Bundesministerium für Land-und Forstwirtschaft, Umwelt und Wasserwirtschaft, Wasserwirtschaftskataster. http://publikationen.lebensministerium.at/

Schmutz S, Zauner G, Eberstaller J (2001) Die "Streifenbefischungsmethode": Eine Methode zur Quantifizierung von Fischbeständen mittelgroßer Fließgewässer. Österr Fischerei 54:14–27

Schmutz S, Matulla C, Melcher A, Gerersdorfer T, Haas P, Formayer H (2004) Beurteilung der Auswirkungen möglicher

Klimaänderungen auf die Fischfauna anhand ausgewählter Fließgewässer. Final report by government department BMLFUW, GZ 54 3895/163-V/4/03. http://www.boku.ac.at/imp/klima/Literatur/FishClim_Endbericht.pdf

Shuter BJ, Post JR (1990) Climate, population variability, and the zoogeography of temperate fishes. Trans Am Fish Soc 119:314–336

Singh VP (1995) Computer models of watershed hydrology. Water Resources Publications, Highlands Ranch, CO

Thienemann A (1925) Die Binnengewässer Mitteleuropas. Eine limnologische Einführung. Schweitzerbart'sche, Stuttgart

Vannote RL, Minshall GW, Cummins KW, Sedell JR, Cushing CE (1980) The river continum concept. Can J Fish Aquat Sci 37:130–137

Welcomme RL (1985) River fisheries. FAO Fisheries Technical Paper (262). FAO–UNO, Rome, http://www.fao.org/documents/show_cdr.asp?url_file=/DOCREP/003/T0537E/T0537E00.HTM

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