Selection of the most appropriate sampling technique and compilation of a common data set as a basis for standardizing a fish-based index between three European countries

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ABSTRACT. — The project intended to adapt and standardize to the European ichthyofauna a new index of ecological quality assessment based on the attributes of fish communities, in order to evaluate the global quality, the conservation and restoration of lotic ecosystems in an international river basin (The Meuse). The methodology developed during this project could be used as a tool for the evaluation of the ecological status of water bodies, in close conformity with the recent European Water Framework Directive (WFD). In this aim, two new indices based on the two major approaches, namely the Index of Biotic Integrity methodology previously adapted for Belgium rivers and the Fish-Based Index methodology previously developed for French rivers were developed and the respective performances of these two newly developed indices are compared.

1. Introduction

In Europe, water policy is currently subjected to considerable change as emphasised by the recent European Water Framework Directive (WFD), which requires the assessment of their hydromorphological, chemical and biological characteristics for the restoration and maintenance of “healthy” aquatic ecosystems. Compared to previous policies, the WFD gives strong priority to biological quality goals by introducing measurements of aquatic biota necessary to identify the structural and
functional integrity of ecosystems. Furthermore, the WFD introduces river basin management throughout Europe, which could have major impacts on the conservation and restoration of aquatic systems (POLLARD & HUXHAM, 1998). Surface water status, according to the WFD, is composed of two elements (i.e. "ecological status" and "chemical status"). Concerning the ecological status, four biological compartments of the ecosystem are taken into account: phytoplankton, macrophytes, benthic invertebrates, and fish communities. All these communities should be characterised by their species richness, species composition, and abundance.

If the requirements of the WFD are to be met, effective biological tools are needed to measure the "health" of rivers at scales large enough to be useful for management (e.g. river basin scale). These tools need to be ecologically based, efficient, rapid and consistently applicable to different ecological regions.

Ecologists have developed biological indices to monitor water quality beginning with the pioneering efforts of KOLKWITZ & MARSSON (1908, 1909). Since this early work, the concept of biological monitoring has been greatly refined with a general trend away from the indicator species concept and/or diversity indices towards an integrated, community-based approach (see FAUSCH et al., 1990 for a review).

For aquatic ecosystems, biological indicators can be chosen from a variety of animal or plant assemblages, but fish are of particular interest because 1) they are present in many water bodies, 2) their taxonomy, ecological requirements and life history are generally better known than for other assemblages, 3) they occupy a variety of trophic levels and habitats, 4) and they have both economic and aesthetic values, and thus help raise awareness about the necessity of conserving aquatic habitats.

There are relatively few suitable ecological tools based on fish communities currently available for assessment of river condition in Europe. Two major approaches (tools) can be distinguished. Both use the "reference condition approach" which involves testing an ecosystem exposed to a potential stress against a reference condition that is unexposed to such a stress.

The first approach (named here as the trisection method index, TMI) to quantify the impact of human activities on the aquatic ecosystem is a multimetric index, the Index of Biotic Integrity (IBI), first formulated by KARR (1981) and later refined by KARR et al. (1986) for use in Midwestern USA streams. The IBI is based on the hypothesis that there are predictable relationships between fish assemblage structure and the
### Table 1

**IBI metrics for Midwestern USA streams (from Karr et al., 1986 and Miller et al., 1988)**

<table>
<thead>
<tr>
<th>Category</th>
<th>Scoring Criteria&lt;sup&gt;a&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>5</td>
</tr>
<tr>
<td>Species Richness</td>
<td></td>
</tr>
<tr>
<td>1. Total number of fish species</td>
<td>b</td>
</tr>
<tr>
<td>2. Number of darter species</td>
<td>b</td>
</tr>
<tr>
<td>3. Number of sunfish species</td>
<td>b</td>
</tr>
<tr>
<td>4. Number of sucker species</td>
<td>b</td>
</tr>
<tr>
<td>Habitat guilds</td>
<td></td>
</tr>
<tr>
<td>5. Number of intolerant species</td>
<td>b</td>
</tr>
<tr>
<td>6. % individuals as green sunfish</td>
<td>&lt; 5</td>
</tr>
<tr>
<td>Trophic guilds</td>
<td></td>
</tr>
<tr>
<td>7. % individuals as omnivores</td>
<td>&lt; 20</td>
</tr>
<tr>
<td>8. % individuals as insectivorous cyprinids</td>
<td>&gt; 45</td>
</tr>
<tr>
<td>9. % individuals as piscivores</td>
<td>&gt; 5</td>
</tr>
<tr>
<td>Abundance</td>
<td></td>
</tr>
<tr>
<td>10. Number of individuals</td>
<td>b</td>
</tr>
<tr>
<td>Reproduction &amp; Condition</td>
<td></td>
</tr>
<tr>
<td>11. % individuals as hybrids</td>
<td>0</td>
</tr>
<tr>
<td>12. % individuals with anomalies&lt;sup&gt;e&lt;/sup&gt;</td>
<td>0-2</td>
</tr>
</tbody>
</table>

<sup>a</sup> Value approximates (5), deviates somewhat (3), or deviates strongly (1) from the reference condition.

<sup>b</sup> Expected value varies with stream size, region, and basin.

<sup>c</sup> Adult diets typically include $\approx 25\%$ plant and $\approx 25\%$ animal material.

<sup>d</sup> Adult diets usually composed largely of aquatic vertebrates or crayfish.

<sup>e</sup> Disease, eroded fins, lesions, tumours, discoloration, excessive mucous, skeletal abnormalities, missing organs, and other external symptoms.

The IBI employs a series of metrics based on assemblage structure that give reliable signals of river condition to calculate an index score at a site, which is then compared to the score expected at an unimpaired comparable site. Classes of metrics in the IBI include species richness, species composition, trophic structure, total fish abundance, and individual fish condition (Table 1). Each metric reflects the quality of a specific aspect of the fish assemblage that responds in a different manner to aquatic ecosystem stressors (Hughes & Noss, 1992). The combination of metrics reflects insights from individual, population, assemblage, ecosystem and zoogeographic perspectives. The IBI methodology is outlined in Table 2. Since its introduction, the IBI has been modified for use in other regions and types of ecosystems throughout North America (see Karr & Chu, 1999 for a review). It has also been modified for use outside North America.
Table 2

Principles of fish assemblage assessment with the IBI

1. Select a relatively homogeneous region. A region may be an eco-region, basin, or fish faunal region that is homogeneous with respect to a combination of environmental characteristics (e.g., climate, physiography, soil, vegetation) and potential fish species.

2. Determine the reference condition(s). References may be a set of minimally disturbed reference streams, a disturbance gradient, historical data, paleoecological information, and professional judgement. Expectations will likely differ for water body size, gradient, temperature or other naturally limiting variables.

3. List candidate metrics and assign species to trophic, tolerance, and habitat guilds. Regional fish texts usually provide this information, at least in developed countries.

4. Sample fish assemblages. This is best done (a) when they are least variable yet most limited by anthropogenic stressors and (b) in a manner yielding a representative collection of species and proportionate abundance, but that (c) is cost-effective.

5. Tabulate numbers of individuals collected by species. Also, determine the total number of individuals collected at each reach.

6. Calculate values for each candidate IBI metric. Typically these are proportions or percents of individuals, or numbers of species in particular categories.

7. Develop scoring criteria. These are based on previously available information from step 2 or from fish data collected at minimally disturbed sites in step 4. Scoring criteria may be continuous (0-1 or 0-10) or based on classes (1, 3, 5 or 0, 5, 10).

8. Calculate metric scores and add these to obtain an IBI score.

9. Evaluate metric and index scores. Consider differences between expected and obtained scores, compare variance results from repeated samples, assess responsiveness to environmental stressors. Modify or reject metrics that are highly variable or unresponsive, and recalculate if necessary.

10. Interpret IBI score as indicating an acceptable, marginally impaired, or highly impaired fish assemblage; or as excellent, good, fair, poor, and very poor.

(see Hughes & Oberdorff, 1998 for a review) and notably in Europe (Oberdorff & Hughes, 1992; Oberdorff & Porcher, 1994; Didier, 1997; Belliard et al., 1999; Belaire et al., 2000; Kesminas & Virbickas, 2000, Kestemont et al., 2000). These applications have shown that the IBI concept is widely adaptable, but that metrics must be modified, deleted, or added to reflect regional differences in fish distribution and assemblage structure (Hughes & Oberdorff, 1998). The use of the IBI in a variety of stream and river ecosystems and in a diversity of geographic areas attests to the utility of the concept (Karr & Chu, 1999).

The second approach (named here the multivariate model index, MMI) originates from a research programme (1996-2000) initiated by the French Water Agencies and the Ministry of the Environment to develop a
fish-based index that would be applicable nation-wide. Such an index had
to encompass the relative importance of geographic, ecoregional and
local factors influencing the distribution of riverine fish. In fact, adapting
such an index over a broad geographic area (i.e., France) required a
detailed understanding of both the patterns of assemblage composition
and distribution within and among water bodies under natural conditions.
and the nature of the major environmental gradients that cause, or at least
explain, these patterns (LYONS, 1996). Patterns of assemblage richness
and composition are strongly influenced by the spatial scale of investiga-
tion. On a local scale (within a site), previous studies of stream fish
assemblages have shown that habitats [as a function of depth, current
velocity, temperature, substrate (HUET, 1959; GORMAN & KARR, 1978;
ANGERMEIER & SCHLOSSER, 1989; RAHEL & HUBERT, 1991)], and increas-
ing stream size [gradient, stream width, discharge, stream-order, catch-
ment area (SHELDON, 1968; HORWITZ, 1978; PALLER, 1994; BELLIARD,
BOET & TALES, 1997)] can influence not only species richness, but also
trophic composition (SCHLOSSER, 1982; ANGERMEIER & KARR, 1983;
ÖBERDORFF et al., 1993). At regional scales (basins or ecoregions), physi-
cal factors such as river size (HUGUENY, 1989; WELCOMME, 1990;
GUÉGAN et al., 1998), geomorphology and climate (HUGHES et al., 1987;
WHITTIER et al., 1988; CHANGEUX & PONT, 1995) are the major determi-
nants of assemblage richness and composition, thereby regulating the
importance of local-scale factors. Thus, before assemblage response can
be useful in the assessment of stream condition and/or in the comparison
of stream condition within an aquatic system (and from one region to
another), using indices such as the IBI, it is essential to take into account
the way assemblage attributes are related to natural stream conditions
(HOEFS & BOYLE, 1995; SMOGOR & ANGERMEIER, 1998). The approach
relies on statistical models (ÖBERDORFF et al., 2001) to predict the site-
specific fauna to be expected in the absence of major environmental
stress. Each prediction requires information on some environmental fea-
tures. The predictions are then compared with the fauna observed at the
similar site, as determined using standardised sampling techniques. The
method uses the amount of deviation between the expected and observed
assemblages within sites as an indicator of the site degradation. A similar
approach has been recently applied to assess environmental disturbance
of Swedish streams (APPLEBERG et al. 2000).

The procedure detailed for the Fish Biotic Index (FBI) from which
MMI was inspired is as follow: (1) A variety of metrics based on occurrence
and relative abundance data and reflecting different aspects of the
fish assemblage structure and function are selected from available literature and for their potential to indicate degradation. (2) Logistic regression procedures are applied, using a data set of reference sites fairly evenly distributed across rivers and defined by some easily measured regional and local characteristics (i.e. hydrographic units, climatic variables, position within the upstream downstream gradient and local habitat characteristics), to elaborate the simplest possible response model that adequately explains the observed patterns of each metric for a given site. The "-response curve-" of a metric describes the most probable value of the metric as a function of a measured environmental variable. (3) After assigning scoring criteria for each metric by analysing distribution values of standardised residuals of each metric model obtained from the first data set of reference sites, models are validated using two independent data sets of reference and disturbed sites. These procedures allow selecting the most effective metrics in discriminating between reference and disturbed sites. The scores for each metric are then summed to produce the final index.

As detailed above, the assessment of environmental degradation using fish communities has received little attention throughout Europe. Thus, in accordance with the Water Framework Directive (WFD) there was an urgent need to develop effective tools based on fish assemblages allowing the objective development of an effective assessment approach of the ecological status of running waters. This was the objective of the present study, which was the first international research initiative (3 countries) in Europe focusing on standardising and adapting a fish-based index for an entire European river basin. Kestemont et al. (2000) and Belpaire et al. (2000) developed an IBI for portions of this basin, but appropriate ecological regions have not been established, nor have metrics with basin-wide utility been identified. To be useful at this scale (Meuse Basin scale), the index must accommodate for potential natural geographic variation in fish assemblages.

2. Material and methods

2.1. Comparison of fishing techniques

The first step was to draw up a list of species of the Meuse Basin and their ecological guilds (Annex 1), a list of potential metrics, and 3 common data files where needed information were compiled (ecological, species and metrics files). During the first 18 months, field activities were
conducted in each country and 5 common fishing operations were performed in order to compare efficiency of each technique in the same site at the same period. Three main techniques were used: electrofishing in wadable rivers and from boat in large rivers, gillnetting and trawling (Fig. 1). The complete description of fishing techniques are presented in Goffaux et al. (2001).

2.2. DATA PROCESSING

As electrofishing data represented the most part of the data set, and for homogeneity reason, it was decided to work only with those data to design the index. The data set contained two kinds of information concerning 698 sites (small and large rivers):
Ecological describers (slope, width, altitude, temperature, conductivity, etc.) including water and habitat quality;

Fish community characteristics (species, abundance, biomass) considered through metrics

This data set was randomly divided into two independent subsets:

- The first one named "design" (351 sites) was used for the calibration stage, defining thresholds and providing scores;
- The second one named "validation" (347 sites) was used to test the efficiency of the two indices.

Among the "design" data set stations presenting the best water and habitat quality (i.e. the least degraded sites) 94 sites were selected as "reference site".

2.3. Selection of environmental factors

- Among several environmental variables 5 were identified as particularly influential on fish community structure:
  - January and July air temperature, reflecting climatic influence;
  - slope;
  - synthetic descriptor (named G) of the position within the longitudinal gradient (distance from source) and the river size (width and watershed area);
  - altitude;
  - one regional variable.

2.4. Selection of metrics

Among the list of potential metrics usually proposed in the scientific literature, 11 metrics were selected for both indices. They were chosen on the basis of their ability to reflect various levels of degradation (Table 3).

2.5. Design and comparison of two approaches

Trisection method (TMI). — This method used the whole design data set (either degraded or undisturbed sites) and two describers of the longitudinal gradient (Watershed area WSA and Altitude Alt). Indeed both describers transformed by classes were considered as the most synthetic parameters to reflect the variations of ecological conditions. Based on the highest value for each class of log WSA or log Alt, the best-fitted curve (e.g. \( y = ax^2 + bx + c \)) was adjusted to describe relationships between the
A FISH-BASED INDEX

Table 3

<table>
<thead>
<tr>
<th>Metric description</th>
<th>Metric code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Species richness</td>
<td>NbSp</td>
</tr>
<tr>
<td>Total number of fish caught per unit effort (100 m⁻¹)</td>
<td>Eff</td>
</tr>
<tr>
<td>Total biomass of fish caught per unit effort (100 m⁻¹)</td>
<td>Biom</td>
</tr>
<tr>
<td>Percentage of lithophilous species minus exotic and tolerant</td>
<td>%Sp Litho</td>
</tr>
<tr>
<td>Percentage of rheophilous species</td>
<td>%Sp Reo</td>
</tr>
<tr>
<td>Percentage of intolerant species</td>
<td>%Sp Int</td>
</tr>
<tr>
<td>Percentage of tolerant species</td>
<td>%Sp Tol</td>
</tr>
<tr>
<td>Percentage of tolerant individuals</td>
<td>%Ind Tol</td>
</tr>
<tr>
<td>Percentage of intolerant individuals</td>
<td>%Ind Int</td>
</tr>
<tr>
<td>Percentage of insertivorous individuals</td>
<td>%Ind Inv</td>
</tr>
<tr>
<td>Percentage of omnivorous individuals</td>
<td>%Ind Omn</td>
</tr>
</tbody>
</table>

Metrics and the WSA or Alt. Then the 1/3 and 2/3 curves were calculated from the previous one. These 3 curves were used to define the thresholds in order to attribute scores (1, 3 or 5) for each metric depending on the expected variation of the metric with degradation.

**Multiple regression method (MMI).** — The second method started with the selection of the least degraded sites considered as “reference” sites. Multiple linear regression was used to modelize each metric as a function of several geomorphologic parameters supposed to be highly influential on fish community structure. It leads to the prediction of the theoretical values for each metric at a given site when no degradation occurs. This allows the deviation between theoretical and observed values to be calculated. Using the design data set, these deviation values were sorted and this range of values was divided into three segments (scores 1, 3 and 5) according to the expected evolution of the metric with degradation.

3. Results and discussion

Both indices (TMI and MMI) were very efficient in discriminating over a gradient of anthropogenic perturbations and the overall proportion of presumed errors of classification was roughly the same for both indices (Fig. 2). TMI and MMI can be considered as broadly based ecological indices that assess fish assemblage structure and function at several trophic levels. They are flexible and widely adaptable. Metrics seem sensitive to many types of degradation, including water and habitat degradations. But contrary to MMI, TMI does not implicitly integrate all
major environmental factors that cause, or at least explain, the patterns of assemblage composition and distribution within and among water bodies at various spatio-temporal scales under natural conditions. These restrictions make the process of establishing appropriate, sensitive metric expectations difficult. Concerning the ability of the two proposed indices to reflect the impact of known habitat and/or water quality perturbations, we observe a linear increase of the scores variability for both indices with assemblage variability, low-quality sites being more variable through time than higher ones (Fig. 3). Whereas the MMI presented a continuous increase of the fish index scores from class 1 to class 5, the TMI displayed a regular increase of the fish index from class 1 to 4 but with a lightly decrease between class 4 and 5. Both indices seem consistent over time and efficient in discriminating degradations. The TMI systematically underscores the quality of rivers, which are usually considered as low or medium quality rivers. This discrepancy concerns more specifically the Flemish sites and large rivers ("bream zone") and to a lesser extent ("barbel zone") with the higher number of errors concerning the least impacted rivers (classes 4 and 5) (Fig. 4).

Another major concern is related to the absolute quality level of reference sites in the Flemish region. Since the quality of the main river is relatively low, the selection of reference site can be debated. However TMI has a good ability in assessing small rivers and sites from the Walloon
Fig. 3. — General distribution of the quality classes for TMI and MMI

Fig. 4. — Distribution of errors (absolute values) for the four regions (France, Wallonia, Flanders, The Netherlands).
region. Concerning MMI no significant differences between the distribution of fish integrity classes and global ecological quality classes were found. The main deviations (even if not statistically significant) relate to a tendency in underscoring small rivers (trout zone) and overscoring rivers from Flanders with the greater proportion of errors concerning classes 2 and 4.

MMI seems to be the more appropriate index for an application to the whole river Meuse basin as it better accounted for regional singularities. Nevertheless, the application of this index requires the selection of reference sites from water quality and, particularly, from habitat quality criteria, with the involvement of appropriate experts advice and/or the application of a well-defined widely accepted and standardised index of habitat quality (which is lacking at this moment). The TMI has overall a good ability in assessing anthropogenic perturbations but tends to underscore systematically the ecological quality of sites and more especially the Flemish sites and large river sites. In the frame of an international application the MMI could be recommended, after further validation in other European river basins, while, the TMI could be used in a regional context. Considering the significant effect of sampling methodology, these indices should be applied exclusively for electrofishing data.

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