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A review of existing fish assemblage indicators and methodologies

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Abstract Fish assemblage indicators developed throughout the world were reviewed and key differences in methodologies depending on ecoregions, basins and contrasting fish fauna summarised. Common elements of existing Indices of Integrity were identified to support the development of a European-wide fish index. These include, using reference condition, accounting for natural fish assemblage variability, evaluating metric precision and selecting the most sensitive and complementary metrics. For future developments, it was recommended to pay more attention to temporal variability in fish assemblages, age structure of key (sentinel) species and fish migration. Testing hypotheses at different steps of the process seems to be the appropriate way to design fish indices.

KEYWORDS: biological assessment, ecological integrity, fish assemblages, IBI, rivers.

Introduction

Monitoring programmes for assessing human impact on river ecosystems have existed for decades. Initially based exclusively on chemical indicators, they increasingly include quantitative biological indicators. The European Union's Water Framework Directive (WFD; EU 2000) underlined the central role of biological indicators to assess the ecological status of rivers. Assessing the biological condition of a site, consists of judging to what extent biotic assemblages collected at the site, conform to those expected under natural or near-natural conditions. To formalise the biological knowledge of experts and to standardise bio-assessment, biological indices are scored proportional to the degree of modification a site's biota has experienced.

Fish are an important element for such assessments because of their biological and socioeconomic status. Consequently, different fish-based indices have been developed worldwide for assessing the ecological status of rivers. Most indices incorporated a reference condition approach and relevant biological variables or metrics (see Noble *et al*. 2007), to describe the fish assemblage characteristics and to quantify the impact of human activities on the biota. The Index of Biotic Integrity (IBI) is the generic name retained to describe this general framework after the work of Karr (1981) and Karr, Fausch, Angermeier, Yant & Schlosser (1986).

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Prior to any development, this approach requires: (1) identification and characterisation of river types; (2) description of the reference fish assemblage for each river type and (3) selection of appropriate biological attributes of the fish assemblage (metrics) that will be used to quantify the difference between observed and reference fish assemblage.

The objectives are to review methods used for designing a fish-based index with a particular focus on original developments in North America and adaptations in European countries and summarise the most common principles applied. The different methods used to assess the ecological status of rivers based on their fish assemblages are compared, focusing on key issues surrounding the reference condition approach, fish assemblage variability (both natural and human-induced) and quantification of the deviation between observed and reference fish assemblages in relation to human disturbance.

Reviewing existing fish-based methods

A variety of quantitative indices have been used to define specific biocriteria, but the IBI (Karr 1981) has been widely applied to fish assemblage data to assess the environmental quality of aquatic habitats. There are many versions of the IBI, as its characteristics change by region and habitat type, but they are all Multimetric Indices (MMI) reflecting important components of community ecology: taxonomic richness, habitat and trophic guild composition, individual health, and abundance. Most use the reference condition approach (Hughes, Larsen & Omernik 1986; Hughes 1995; Bailey, Kennedy, Dervish & Taylor 1998), which involves testing an ecosystem exposed to a potential stress against a reference condition that is in natural or near-natural state. Angermeier, Smogor & Stauffer (2000) classified these biological indicators as MMI. Such MMIs have been developed for all continents except Antarctica (see Hughes & Oberdorff 1999).

Historical background: the original IBI

As initially proposed by Karr (1981), the IBI is a practical tool to classify sites by ranking them over a range of biotic condition from no fish and poor sites to excellent ones. The originality of the method relies on the descriptors used for that classification and the way they are combined within a final index. These criteria consisted of several complementary attributes of fish assemblage condition that account for various aspects of fish status from individual, guild and assemblage levels. Thus, this approach constituted a straightforward method compared with diversity indices, indicator species, species lists or multivariate analyses, as it seeks a more holistic, integrative and ecological approach.

Karr (1981) proposed a rating system for each metric, but the principle remained vague. However, the fundamentals were established to assign a score ranging from 1 to 5 for each metric, depending on the strength of the deviation from the excellent expected scenario at reference sites (Table 1). Even if thresholds allowing discrimination between levels of biological condition were established, no practical rule was described. Due to natural variability of fish assemblages, the difficulty in strictly defining a reference baseline was underlined, but no alternative to circumvent that problem, and no option to account for the

Table 1. Metrics and scoring criteria of the original IBI (from Karr *et al.* 1986; Miller *et al.* 1988)

Category	Metrics	Scoring criteria 5	3	1
Species richness	Total number of fish species		Varies with stream order	
Species composition	Number of darter species		Varies with stream order	
	Number of sunfish species		Varies with stream order	
	Number of sucker species		Varies with stream order	
Tolerance guilds	Number of intolerant species		Varies with stream order	
	% Individuals as green sunfish	< 5	5–20	> 20
Trophic guilds	% Individuals as omnivores	< 20	20–45	> 45
	% Individuals as insectivorous cyprinids	> 45	20–45	< 20
	% Individuals as piscivores	> 5	1–5	< 1
Abundance	Number of individuals		Varies with stream order	
Reproduction and condition	% Individuals as hybrids	0	> 0–1	> 1
	% Individuals with anomalies	0–2	> 2–5	> 5

Scoring: 5, near reference; 3, significant deviation from reference; 1, strong deviation from reference.

longitudinal zonal changes of fish assemblages in rivers was proposed. Moreover, very little information was given on the types of rivers considered and the presumed level of ecological condition of sites, nor on the way metrics were selected. While stating the rationale of the IBI method, Karr (1981) focused on the question of representativeness of fish samples with respect to both fish assemblage and river reach.

Details on the application of the IBI method were first given by Fausch, Karr & Yant (1984). They offered explanations concerning the scoring system and paid particular attention to the influence of river size and regional factors. Karr *et al.* (1986) further developed the rationale and the practical rules of the method to allow wider applications. Since these initial proposals, a considerable number of applications have been made worldwide, from which numerous questions have arisen, in relation to the biological, ecological, statistical or practical features of fish-based indices for assessing ecological condition of rivers.

Defining a theoretical fish assemblage: 'type-specific' vs 'site-specific' approaches

Most predictive models assess biological status by comparing the biotic condition at sites being evaluated, with the biota expected in the absence of human impact (Wright 1995). Of the numerous techniques used, two general approaches (differing in the basis of their analyses) have emerged. The type-specific approach relies on clustering techniques as a starting point, so that the reference sites are classified into groups based on ecoregions or faunal homogeneity. This first step, prior to modelling aquatic assemblages, is common to many biomonitoring programmes (Wright, Sutcliffe & Furse 2000). The site-specific approach, does not require prior development of a classification system, and directly predicts expected fauna according to environmental features of the evaluated site (e.g. Oberdorff, Pont, Hugueny & Chessel 2001). Surprisingly, there are few comparisons of these two approaches in the literature (e.g. Goffaux, Roset, Breine & De Leuw 2001 on the Meuse basin) and most IBIs use type classification.

There are drawbacks with both approaches (e.g. Chessman 1999; Olden 2003). The site-specific approach is most appropriate where biological indicators show a continuum of variation, whereas site pre-classification is most appropriate where species or assemblage descriptors exhibit a discrete organisation pattern. Therefore, the reason why one approach is chosen should be justified, and examining patterns of spatial variability is crucial before embarking on a given assessment procedure.

An empirical approach based on fish assemblage conditions

In virtually every regional application of the IBI, evaluation is related to a reference condition (Hughes *et al.* 1986; Hughes 1995; Bailey *et al.* 1998). In practice, however, how the reference condition is defined is not clear and the criteria considered vary greatly. Implicit in the initial IBI, reference is an empirical concept based on potential attributes (metrics) of fish assemblages against which a given sample of the same river type within a region can be compared (Karr *et al.* 1986). In many cases, however, the threshold values for a metric (or a site) to be judged as high quality are directly assigned by expert opinion, on the basis of biological background and personal observations (Karr 1981; Lyons, Navarro-Pérez, Cochran, Santana & Guzman-Arroyo 1995; Hugueny, Camara, Samoura & Magassouba 1996). The reference is usually the maximum value recorded for a given metric, which is a less subjective approach (Fausch *et al.* 1984; Oberdorff & Hughes 1992; Belpaire, Smolders, Vanden Auweele, Ercken, Breine, Van Thuyne & Ollevier 2000; Kestemont, Didier, Depiereux & Micha 2000; Schleiger 2000).

In all these regional applications, the reference should be viewed as the least-impacted sites within a biogeographical region (Hughes 1995), rather than an absolute or pristine condition, because real pristine conditions rarely exist in most industrialised countries. Thus, referring to unrealistic conditions would make restoration goals not attainable (Tejerina-Garro, Maldonado, Ibanez, Pont, Roset & Oberdorff 2005).

Explicit approach based on environmental evaluation

Another way to use the reference condition involves selecting the sites that exhibit the best environmental status for most physical and chemical criteria. Based on these restricted sites, the fish assemblage attributes are described and serve as references for assigning scoring criteria. This approach has been rarely used until recently (Belpaire *et al.* 2000; Lyons, Gutierrez-Hernandez, Diaz-Pardo, Soto-Galera, Medina-Nava & Pineda-Lopez 2000; McCormick, Hughes, Kaufmann, Peck, Stoddard & Herlihy 2001; Oberdorff *et al.* 2001; Oberdorff, Pont, Hugueny & Porcher 2002), and it satisfies the definitions of both high biotic condition

(Karr & Dudley 1981) and reference as defined by Reynoldson, Norris, Resh, Day & Rosenberg (1997).

Using historical data instead of minimally disturbed sites

Most IBI applications are based on existing data or specially designed and appropriate monitoring programmes. Reference sites are sometimes considered to be pristine, but typically correspond to the least-impacted sites of the study region (Hughes 1995). However, when no reference site exists, some authors (e.g. Schmutz, Kaufmann, Vogel, Jungwirth & Muhar 2000), advocated the use of historical data. For example, Hughes, Kaufmann, Herlihy, Kincaid, Reynolds & Larsen (1998) used Pre-columbian characteristics as reference, while Kestemont *et al.* (2000) and Bryce, Hughes & Kaufmann (2002) accounted for the presumed occurrence of extinct or extirpated species within a catchment. Such approaches are useful in that they avoid integrating existing impacts but their ecological significance is questionable, as it does not account for the natural evolution of these sites. In other words, historical references may not be comparable with the current situation as they might exclude natural evolution that may have occurred even when no degradation occurred. Moreover, the lack of quantitative information in historical data affects the reliability of historical references, particularly for the site-specific approach.

Considerations in sampling

As with most ecological studies, biological assessment requires a consistent and cost-effective method to sample the fish assemblage (i.e. representative of the studied reach) (Hughes, Kaufmann, Herlihy, Intelmann, Corbett, Arbogast & Hjort 2002). This is particularly important because sampling efficiency and sampling effort strongly influence the IBI scores (Angermeier & Karr 1986; Simon & Sanders 1999; Reynolds, Herlihy, Kaufmann, Gregory & Hughes 2003). Thus, key questions to address when developing a fish index concern the delineation of the sampling area, the minimum sampling effort and the sampling strategy.

Electric fishing is the most commonly employed fish sampling method in rivers (Cowx 1995) as it is effective for a wide range of river types (Simon & Sanders 1999), although the use of seines, gill nets or trawls or a combination of several gears is sometimes necessary, particularly for large rivers (Simon & Sanders 1999). However, Simon & Lyons (1995) underlined the problems caused by aggregating data resulting from different gears and sampling efforts within the IBI.

Several authors observed significant effects of sampling effort on IBI scores, due to the higher species richness in high-effort samples (Angermeier & Karr 1986; Reynolds *et al.* 2003). Thus, fish samples should be large enough (Angermeier *et al.* 2000 recommend to capture at least 50 fishes) and must faithfully reflect the species richness and be representative of the relative abundance of the different species present (Fausch *et al.* 1984; Karr *et al.* 1986). In addition, the fish sampling sites should be representative of the whole river reach they belong to, and in particular they must encompass several riffle-pool sequences (or meander wavelength) and represent the variety of habitat types (for example Fausch *et al.* 1984; Angermeier & Karr 1986; Lyons *et al.* 1995; Langdon 2001; Hughes *et al.* 2002; Reynolds *et al.* 2003). As a rule, a minimum sampling distance (and/or sampling time) is often given as a function of wetted width, but this criterion is highly variable (generally from 10 to 40 times the wetted width, up to 198 times wetted width where boats are used). While the prescribed distance to sample generally ranges between 80 and 200 m, it can reach several kilometres in the case of large rivers. In some studies, the distance necessary for species richness to asymptote is retained as the criteria (Simon & Sanders 1999; Reynolds *et al.* 2003).

Another key-point is the number of fishing passes required for small- and medium-sized rivers. In general, several passes are recommended for a statistical estimation of fish abundance and population structure, but the number is highly variable (from 1 to 7). However, in terms of effort, sampling a large area with one pass is considered more efficient than sampling a smaller area with many passes (Paller 1995).

As stressed by several authors (Simon & Emery 1995; Reash 1999; Simon & Sanders 1999; Reynolds *et al.* 2003), these questions concerning sampling are of particular concern for large rivers, where complete inventory of the sampling reach is not possible and the use of a boat is often required.

Similar questions arise concerning rare species (Yoder & Rankin 1995; Oberdorff *et al.* 2001), non-native species or migratory species. For rare species, Simon & Sanders (1999) indicated that the purpose of most studies is not to collect every species but rather to document a representative sample of the fish assemblages. Furthermore, Yoder & Rankin (1995) stressed that failure to collect a few rare species at a site would not detract the assessment because of their very low contribution to the total IBI score. In addition, sites with non-resident species (migratory species or species dispersing from reservoirs) are sometimes excluded from reference conditions (Karr *et al.* 1986) because the probability of capturing these species varies with season. This is questionable as these migratory species are also included in some IBIs as indicators of effective connectivity, a significant aspect of ecological condition.

Karr (1981) also mentioned the problem of young-of-the-year variations that could disrupt assessment, a problem addressed by several authors (Angermeier & Karr 1986; Angermeier & Schlosser 1987). To overcome this problem, Shields, Knight & Cooper (1995), for example, recorded only fish longer than 2 cm.

Identifying the main sources of variability

Large-scale spatial patterns

Omernik (1987), Hughes, Rexstad & Bond (1987) and Whittier, Hughes & Larsen (1988) examined the extent to which ecoregion, river basin, stream order and river size could influence fish assemblage structure and composition. They recognised that these factors could highly modify metric responsiveness and IBI scores (see for example Tejerina-Garro *et al.* 2005 for a review of biotic and abiotic factors related to IBI development). As ecoregion plays a central role in determining the pool of species likely to be resident in a study area, region-specific assessments have become a characteristic of the IBI approach (Fausch *et al.* 1984; Miller, Leonard, Hughes, Karr, Moyle, Schrader, Thompson, Daniels, Fausch, Fitzhugh, Gammon, Halliwell, Angermeier & Orth 1988; Hughes *et al.* 1998; Angermeier *et al.* 2000; Oberdorff *et al.* 2001). However, Angermeier *et al.* (2000) considered river basin to be more influential. Although the relative influence of these two large-scale factors is difficult to assess (Omernik & Bailey 1997), it should be given consideration when developing fish-based indices at large scales (Omernik & Griffith 1991), and is critically an important issue within the EU FAME project (Development, Evaluation and Implementation of a standardised Fish-based Assessment Method for the Ecological Status of European Rivers; http://fame.boku.ac.at) (Schmutz *et al.* 2007; Pont *et al.* 2007).

Similarly, longitudinal changes in species richness and species composition within a river system have been recognised for some considerable time (Huet 1959; Sheldon 1968; Vannote, Minshall, Cummins, Sedell & Cushing 1980) as has the role of abiotic parameters in determining fish assemblage structure (see Matthews 1998). Thus, beyond the usual regional classification, most IBIs usually define a preliminary river classification to account for the natural pattern of fish assemblage organisation. The way these river types are established varies greatly in terms of criteria and the spatial scales considered (i.e. precision). River typologies generally refer to water temperature, river size, slope and position within the drainage network. In the early IBI version (Fausch *et al.* 1984), and several adaptations (e.g. Karr *et al.* 1986; Bramblett & Fausch 1991), stream size was represented by stream order; more recently it has been increasingly replaced by watershed area (Miller *et al.* 1988; Steedman 1988; Oberdorff & Hughes 1992; Harris & Silveira 1999; Kestemont *et al.* 2000; Lyons *et al.* 2000; Schleiger 2000). Other parameters, such as distance from source or distance from confluence (Osborne, Kohler, Bayley, Day, Bertrand, Wiley & Sauer 1992), are also considered. Some authors (Leonard & Orth 1986; Shields *et al.* 1995) used dichotomous classification based on river temperature (cool water vs warm water) instead of regional distinction, while others chose rough separation such as small vs large rivers (Lyons *et al.* 1995; Toham & Teugels 1999). Other classifications are based on combinations of environmental parameters such as river width and slope in the Huet (1959) zonation (e.g. Belpaire *et al.* 2000), or on fish assemblage characteristics highlighted by clustering procedures (Kesminas & Virbickas 2000).

In conclusion, several criteria for river type classification are used in addition to the quasi-systematic regional (and/or river basin) contrast. Beyond the diversity of these parameters, and despite the multiple factors likely to control fish attributes (e.g. altitude, width, temperature), most studies only use one, or rarely two, parameters. This is questionable because the combination of multiple parameters can have a significant influence on fish assemblage structure, response of metrics and IBI variations (Belpaire *et al.* 2000; Lyons *et al.* 2000; Oberdorff *et al.* 2001; Joy & Death 2002). Furthermore, a significant part of the natural variability remains in the index, which could hamper sensitivity of the index to human disturbance.

Temporal variability

Similar to spatial variability, a reliable assessment of biotic integrity requires quantification of natural, temporal variability in fish assemblage structure. An ideal way would be to use a reference data in which each site is sampled in the same season, for at least 5 yr (minimum duration necessary for collecting 96% of the total cumulative number of species according to Simon & Sanders 1999). However, no study has accounted for temporal variability explicitly and consistently. At best, some studies include year-to-year replicates, but the number of replicates is variable from one site to another, leading to an unbalanced data set. Similarly, seasonal variation is never rigorously accounted for in the reference data set.

Instead, greater attention has been paid to temporal variability of indices and their component metrics (e.g. Hughes *et al*. 1998; Harris & Silveira 1999; Bozzetti & Schulz 2004). They found that temporal variability was lower than between site variability. Furthermore, Paller (2002) observed a greater persistence and stability of fish assemblages and less variation of IBI scores over time at undisturbed sites and concluded that assessment of temporal variation in fish assemblage structure could serve as an indicator of environmental disturbance.

Natural vs anthropogenic variability

The implications of accounting for ecosystem variability in the frame of biological assessment have been progressively outlined. Karr *et al*. (1986) first stressed the need to define the range of natural variability in stream assemblages, so that techniques to distinguish natural from anthropogenic variations can be developed. They emphasised the need to not fix natural condition but rather understand more accurately the patterns of natural variation. Indeed, abiotic descriptors can explain fish assemblage variations through two distinct processes: natural environmental fluctuations and anthropogenic disturbance. As emphasised by Fausch, Lyons, Karr & Angermeier (1990), a central problem in biological assessment is '*the documentation of natural variation in fish communities against which changes due to degradation can be compared*'. Thus, as pointed out by Fore, Karr & Conquest (1994), ecological realism, sensitivity and statistical precision of biological tools for assessing impacts of human activities on river biota, strictly depend upon our ability to distinguish between natural and human-induced processes.

Since then, three main solutions to account for natural variability have been proposed implicitly in most IBI applications. The first consists of defining discrete ichthyo-regions and/or river types and assigning different assessment criteria according to that typology. The second involves representing, in most cases graphically, the relationships between a given metric and a descriptor of spatial variation supposed to be among the main factor influencing fish assemblage organisation (e.g. river size, position within the longitudinal gradient). As an extension, a third consists of modelling metric values with a set of environmental factors as explanatory variables (Oberdorff *et al*. 2001). Initially, these relationships were estimated visually and studied only for a restricted number of metrics, generally those representing species richness characteristics (Fausch *et al*. 1984; Karr *et al*. 1986).

More recently, metric response to environmental conditions has been statistically tested using models of varying complexity, and in which river types and degradation effects were sometimes, but not always, included as distinct factors (Lyons *et al*. 1995, 2000; Harris & Silveira 1999; Toham & Teugels 1999; Belpaire *et al*. 2000). In some cases, spatial variability is considered for all metrics (Ganasan & Hughes 1998; Kestemont *et al*. 2000; Schleiger 2000; Oberdorff *et al*. 2001).

In many cases, scoring criteria for a given metric are established along a range of river sizes using a graphical procedure: the Maximum Value Line (MVL) principle, first introduced by Fausch *et al*. (1984), and then generally applied with modifications. Two methods have been developed to establish the MVL: (1) the line, which encloses 95% of the samples, establishes the maximum likely values of the metric for a given size group within a given region and (2) the highest metric values usually obtained at the least disturbed sites are used as estimates of the reference condition (e.g. Kestemont *et al*. 2000) (Fig. 1). In both cases, linear or second order polynomial regression curves have been used, depending on how accurately the regression curve fitted the data. The area between this line and the x-axis is then divided into a number of segments. Each segment is assigned a score, depending on whether the deviation from the expected situation is high (degraded situation) or low (little degradation

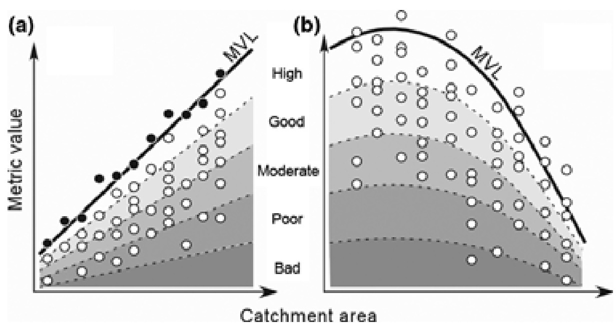

Figure 1. Illustration of the maximum value line (MVL) principle using the relationships of two hypothetical metrics with catchment area. Dots represent values for sampling sites. MVL was established (a) using the highest metric values (black dots) or (b) enclosing 95% of the samples. The area below the MVL was divided into five sections and used to rate the corresponding metrics as indicating high, good, moderate, poor or bad status.

occurred). Thus, the upper or lower segments represent the least disturbed situations, depending on the expected response of the metric to degradation. Indeed, each metric contributes to a fish-based index by considering its probable variation with degradation (see Noble *et al*. 2007). Typically, metric values can increase (e.g. the number of omnivorous species), decrease (for example the density of sensitive species) or both (for example species richness that could increase with limited nutrient load and decrease following toxic or organic pollution).

The differences observed in the various adaptations of this procedure mainly concern the number of categories retained. Most authors use the classical three-level segregation (trisection), while others use five categories (Belpaire *et al*. 2000; Kestemont *et al*. 2000) or a continuous rating scale (Minns, Cairns, Randall & Moore 1994; Ganasan & Hughes 1998; Hughes *et al*. 1998; McCormick *et al*. 2001). Similarly, different rating levels are used for each category. In preference to the usual 1/3/5 system, some use 1/2/3 (Simon & Emery 1995), 0/5/10 (Lyons *et al*. 1995, 2000) or 1/2/3/4/5 (Belpaire *et al*. 2000).

Other differences pertain to the techniques used to relate metric trends to river size or to how the maximum line is adjusted to site responses. Although it was initially fitted by eye, it is sometimes fitted through the use of linear regression models (Didier & Kestemont 1996; Belpaire *et al*. 2000), which usually requires using only pre-selected reference sites (Lyons *et al*. 2000; Oberdorff *et al*. 2001). Both alternatives are often used within the same index, depending notably on the metric and the link with river size or other river type variables (i.e. the existence of a particular trend of metric values with river type, and the strength of this relationship). However, in many cases, the natural variability of the metric is totally neglected and a fixed threshold value is assigned for a given river type.

Recently Oberdorff *et al*. (2001) proposed a singular way of dealing with natural variability and scoring metrics. The score of a metric is based on the probability for an observed value of this metric to belong to the distribution of reference values. To remove first the natural variability, they proposed to use the residual of the general linear models.

Reviewing existing fish-based methods in Europe

This section reviews the methods developed in different European regions or countries to assess the ecological quality of rivers by using fish-based methods. It is based on national methods developed in France (Oberdorff & Hughes 1992; Oberdorff & Porcher 1994; Oberdorff *et al*. 2001), Belgium (Belpaire *et al*. 2000; Kestemont *et al*. 2000; Breine, Simoens, Goethals, Quataert, Ercken, Van Liefferinghe & Belpaire 2004), Austria (Schmutz *et al*. 2000), Lithuania (Kesminas & Virbickas 2000), Sweden (Appelberg, Berquist & Degerman 2000), the UK (Rahman 2002) and a first European initiative covering an international river basin (Goffaux *et al*. 2001).

Despite the indices being developed independently, all methods have retained the original ecological framework, including a series of metrics that describe the major biological attributes of a fish assemblage. Some IBI methods have been developed for regional applications, or for specific river types (such as small headwaters with poor species diversity), while others aim to be applied to a wider area, an entire ecoregion, or even several ecoregions with different ichthyofaunas. The aim of this section is not to describe in detail the existing methods, published elsewhere, but to point out the common characteristics among the various indices and to underline their development through successive adaptations.

Comparison of these main characteristics is summarised in Table 2. Most existing fish-based methods developed in Europe assess ecological condition of lotic waters from several large river basins, even if some are particularly adapted for a certain part of the upstream–downstream gradient. Apart from two methods (Kesminas & Virbickas 2000; Schmutz *et al*. 2000), all others mainly follow a site-specific approach to predict expected fish fauna, even if some of them also take into account river types as a prerequisite for metric selection and/or scoring. It is notable that most existing methods consider natural spatial variation in fish assemblages by using one spatial descriptor only (i.e. MVL principle), while methods using more complex models (e.g. Goffaux *et al*. 2001; Oberdorff *et al*. 2002) remain poorly developed. The three- or five-level rating scale for metric scoring is the most commonly used, whereas the way authors examine metric contribution to the final index shows greater heterogeneity among the various screened methods.

Overview of metrics used for IBI development worldwide

Since the first version of the IBI in North America (Karr 1981), the IBI concept has been tested or adapted throughout the world (for review, see Miller *et al*. 1988; Simon and Lyons 1995; Hughes & Oberdorff 1999). Despite a huge diversity in metrics chosen worldwide (Noble *et al*. 2007), most IBIs have

Table 2. Comparison of key characteristics of IBIs adapted to European rivers (cross referencing of use of metric in models by authors listed in columns)

Category	Items		1	2	3	4	5	6	7	8	9	10	11
Spatial scale of assessment	River size	Headwaters	X	X	X	X		X	X	X	X		X
		Medium streams	X		X	X	X		X	X	X	X	X
		Large rivers	X		X		X		X	X	X	X	X
	Biogeographic area	Single large hydrographic unit, ex. Seine or Meuse…	X			X					X		
		Multiple large hydrographic unit, ex. Seine + Meuse + Rhone + Loire		X	X		X	X	X	X		X	X
Characterising reference conditions	Principle	Site-specific approach	X	X	X	X	X	X	X		X	X	
		Type-specific approach					X			X			X
	Method for natural variability account	Standardised criteria for river classification								X		X	X
		Maximum value line fitted by eye	X	X									
		Maximum value line fitted by simple linear regression				X	X	X			X		
		Multiple regression modelling			X						X		
Metric scoring		Three level rating scale	X	X				X			X		
		Five level rating scale				X	X		X			X	X
		Other (2, 3…depending on the metrics)								X			
		Continuous rating scale			X								
Testing metrics sensitivity and redundancy		ANOVA									X		
		Correlation analysis					X	X					
		t-tests (reference vs disturbed sites)			X								
		Multiple linear regression	X	X									
		Principal component analysis				X		X					
		Discriminant analysis									X		

1, 2, 3: France (Oberdorff & Hughes 1992; Oberdorff & Porcher 1994; Oberdorff *et al*. 2002); 4, 5, 6: Belgium (Belpaire *et al*. 2000; Breine *et al*. 2004; Kestemont *et al*. 2000); 7: Sweden (Appelberg *et al*. 2000); 8: Lithuania (Kesminas & Virbickas 2000); 9: International river Meuse (Goffaux *et al*. 2001); 10: UK (Rahman 2002); 11: Austria (Schmutz *et al*. 2000).

included the original classification of metrics among four major categories: species composition and diversity, trophic composition, fish abundance and fish condition.

Grouping the dominant metrics into four main categories (species composition and diversity, trophic composition, fish abundance, reproduction success and requirement), allows the comparison of the mean relative importance (%) of these categories in North American and European IBIs (Fig. 2). The mean relative number of metrics belonging to the trophic group is roughly the same (about 20%), but the relative importance of abundance and reproduction metrics is lower in North American than in European IBIs (7.6% against 16.3%, respectively), while it is the opposite for species composition metrics (55% in North American sample against 41% in Europe).

These contrasts are probably related to the difference of species richness in the continents. Assessing biotic integrity in the species-poor European rivers requires focusing more frequently on abundance metrics and reproduction. For example, in Europe

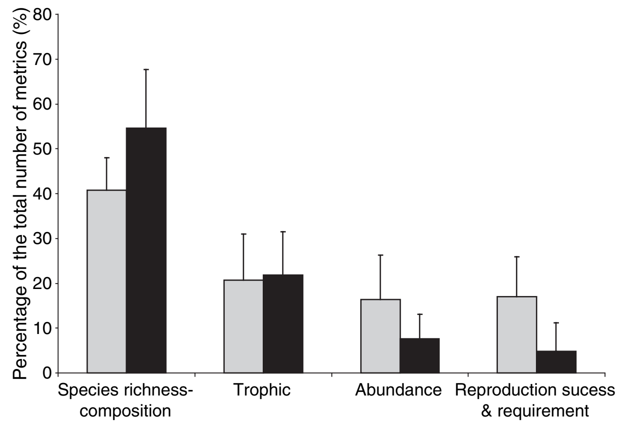

Figure 2. Comparison of the mean (and SE) relative importance (%) of metric categories in European (grey bars) and North-American (black bars) IBIs.

experiences, more attention has been paid to metrics dealing with age or size class distributions for the dominant and/or most intolerant species (e.g. number of year classes for brown trout, *Salmo trutta fario* L. or pike, *Esox lucius* L.). The more recent North American IBIs share some of these evolutions, particularly for coldwater and species-poor streams (Lyons, Wang & Simonson 1996; Mundahl & Simon 1999; Langdon 2001; Hughes, Howlin & Kaufmann 2004). In that river type, the number of metrics used tends to be lower and a greater weight is given to cold stenotherm species metrics in the final index (for example half the metrics belong to that category in Hughes *et al.* 2004). Moreover, beyond their low species richness, the metric response is often singular in that river type, for example species richness is more likely to increase with moderate physicochemical or biological alterations.

Studies using metrics describing migration requirements for fish, and particularly large-scale migration, are scarce (only one example in Europe and one in a recent North American development). The same is observed for longevity metrics that is expected to reflect long-term changes.

In large rivers, European studies often use the same set of metrics as in small streams, but in few studies some specific metrics are proposed (e.g. pike age classes). In North America, most of IBIs were designed historically for small- and medium-sized streams. Only recently, attempts have been made to develop specific index for large or great rivers (Simon & Emery 1995; Reash 1999; Lyons, Piette & Niermeyer 2001). Thus, specific metrics have been proposed such as the percentage of species belonging to large river faunal group, the percentage of round-bodied suckers and the number of typically riverine species. Surprisingly, while longitudinal barriers are among the typical impacts of regulated and impounded large rivers, migration metrics were not tested in these studies.

Metric selection

Few studies conducted a rigorous step of metric selection from a larger list of candidate metrics, and those using objective criteria and statistical procedures are scarce (Hughes *et al.* 1998; Roth, Southerland, Chaillou, Klauda, Kazyak, Stranko, Weisberg, Hall & Morgan 1998; Angermeier *et al.* 2000; Kestemont *et al.* 2000; McCormick *et al.* 2001; Oberdorff *et al.* 2002; Mebane, Maret & Hughes 2003). For most applications, 10–12 metrics are retained *a priori* from expert knowledge or previous studies, to be integrated into a MMI. However, few authors make *a posteriori* tests of their sensitivity, redundancy and consistency; nor do they test the initial hypothesis concerning the expected trends of the metrics with degradations.

When redundancy was tested, the procedures applied were often not very rigorous or complete. Most studies used a correlation index as an indicator of redundancy (Hughes *et al.* 1998, 2004; McCormick *et al.* 2001; Mebane *et al.* 2003), often in combination with sensitivity analysis using ANOVA and/or discriminant analysis. The range of variation of a given metric as a function of degradation type and intensity were rarely measured (i.e. for what kind of degradation and in which range a metric is relevant) (McCormick *et al.* 2001). At least, expected ranges for some groups of metrics were sometimes proposed (Angermeier & Karr 1986).

Sensitivity tests are most frequently run on the total IBI score vs quantitative or qualitative descriptors of anthropogenic degradation, including habitat and/or water quality components (e.g. Karr *et al.* 1986; Leonard & Orth 1986; Angermeier & Schlosser 1987; Karr, Yant, Fausch & Schlosser 1987; Steedman 1988). Among the papers screened, only Angermeier *et al.* (2000) examined accurately: (1) to what extent metrics at high-quality sites varied among basins and among ecoregions and (2) how metrics varied among site-quality classes within each region. Unlike many studies, the statistical criteria to be met for a metric to be integrated into a MMI was described by Hughes *et al.* (1998, 2004), Roth *et al.* (1998); Angermeier *et al.* (2000), McCormick *et al.* (2001) and Oberdorff *et al.* (2002). This type of rule is particularly helpful as it establishes an objective and standardised method for selecting the most relevant (i.e. both sensitive and not redundant) metrics.

Many authors also pointed out that values of some metrics may be masked by other attributes (e.g. the exotic/introduced origin or tolerance status), thus confounding their assumed response to degradation (Oberdorff & Hughes 1992; Angermeier *et al.* 2000; Belpaire *et al.* 2000; Kestemont *et al.* 2000). Consequently, to calculate some metric values (e.g. the number of lithophilic species expected to decrease with degradation), they proposed excluding non-native and/or tolerant species, which interfere with the expected metric response to degradation.

In conclusion, procedures for rigorously selecting the most relevant metrics have been increasingly used in the past 10 yr, but methods for defining the number of metrics to be combined within the final index have rarely been reliably explored. Only Roth *et al.* (1998) proposed an iterative process based on the successive addition of metrics and test of the consecutive classification efficiency of the resulting IBI. He found that the use of only three metrics allowed 85% of sites to be classified accurately, and that adding new metrics did not significantly increase efficiency (87% for five metrics).

Conclusion – future needs

Despite the variety of adaptations throughout the world, the fundamentals of the original IBIs have been maintained: a reference condition approach, a process for describing natural variability of fish assemblages, the selection of appropriate fish attributes to quantify the deviation between observed and theoretical fish assemblages through a standardised rating system. Beyond the differences of fish fauna, our study gave evidence that some synthetic and ecologically sound descriptors of fish community exhibit convergent responses to human disturbance in Europe and in North America. This feature makes it possible to design a fish-based index usable at a large scale such as Europe.

Irrespective of whether IBI methods for assessing biological integrity of rivers and streams based on fish assemblages are used in national or states monitoring programmes, a lot remains to be done before they give complete satisfaction to end-users. Future research should focus on:

- Estimating and integrating consistently the natural temporal variability of fish assemblages when describing or modelling reference conditions.
- Standardising the fish sampling methods at the scale of the assessment, for both reference and study sites.
- Selecting reference sites based on standardised criteria, and include where appropriate reliable historical information.
- Testing new metrics, particularly for species-poor and large rivers, with a special attention to age structure characteristics and migration requirements of fishes.
- Selecting appropriate metrics from a large list of candidate metrics, following statistical procedures including sensitivity and redundancy, and test *a posteriori* the initial working hypothesis, for example the expected variations of metrics with degradations.
- Proposing a method to define objectively the optimum number of metrics to be included in the final index.

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