Revisiting ecological integrity 30 years later: non-native species and the misdiagnosis of freshwater ecosystem health

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Abstract

Assessing the ecological integrity of freshwater ecosystems has become a priority to protect the threatened biodiversity they hold and secure future accessibility to the services they provide. Some of the most widespread applications of biological indicators are fish-based indices. These have mostly mirrored the approach proposed by Karr 30 years ago (Index of Biotic Integrity; IBI), based on the comparison of observed and expected composition and structure of local fish assemblages in the absence of major perturbations, using the so-called reference condition approach. Despite the notable success of the implementation of fish-based indices, most of them overlook non-native species as a source of ecosystem degradation, and evaluations are focused on the physico-chemical condition of freshwater ecosystems and their effects on freshwater biodiversity. Almost 90% of 83 reviewed IBIs did not consider non-native species when defining reference conditions. Most IBIs used non-native species in conjunction with native ones to construct the metrics that conform to the index. The response of the IBI to the effect of non-native species has hardly ever been tested. When developing and evaluating IBIs, attention was mostly directed to ensuring the correct response of the index to physico-chemical parameters, which could otherwise be characterized more effectively using alternative methods. Current application of IBIs entails a misuse of biological indicators by overlooking some types of degradation that cannot be otherwise evaluated by traditional methods. This constrains the capacity to adequately respond to one of the most challenging and common threats to the conservation of freshwater fish diversity.

Keywords Biological indicator, ecosystem management, fish, non-native species, river health, Water Framework Directive.

Biological indicators as a diagnostic tool of river health

Freshwater ecosystems and biodiversity are among the most threatened and modified environments on the planet, because of the intensive human use of water resources (Dudgeon *et al.* 2006). These systems are crucial for human society as a source of water for domestic, agricultural and industrial uses, electricity generation and waste disposal, among others (Malmqvist and Rundle 2002). All these activities have altered the ecology and functioning of freshwater systems, through habitat degradation and fragmentation, physical and chemical alterations, and the introduction of non-native species (Dudgeon *et al.* 2006). In some cases, this degradation threatens the continuity of the services that humans receive from freshwater ecosystems (e.g. Vörösmarty *et al.* 2010). For the same reason, freshwater biodiversity is among the most threatened in the world (Dudgeon *et al.* 2006).

To help resource managers and the general public to understand the causes and ecological consequences of the degradation of running water systems, scientists have applied the concept of health to ecosystems (Karr, 1999; Norris and Thoms 1999). However, different terms have been used to refer to ecosystem health in scientific literature, such as biotic integrity (Karr and Dudley 1981) or ecosystem integrity (Karr *et al.* 1986). To accommodate the terminology used here to the most comprehensible and commonly used, we will refer to ecosystem health as ecosystem integrity hereafter. A living organism is healthy when it is free of physical disease and performs all its vital functions normally and properly, being able to recover from normal stress. Similarly an ecosystem can be considered with good integrity when it has the 'capacity of supporting and maintaining a balanced, integrated, adaptive biological community having a species composition, diversity and functional organization comparable to that of natural habitat of the region' (Karr and Dudley 1981). Ecosystems with good integrity can withstand and recover from natural environmental perturbations (Karr *et al.* 1986). As for living organisms, these ecosystems have different components that are susceptible of being affected by diseases so defining its health. In their original definition of ecosystem health, Karr and Dudley (1981) cited four main components: (i) water quality evaluated by factors such as temperature, dissolved oxygen and turbidity; (ii) habitat structure determined by spatial and temporal complexity of the physical habitat, substrate type and water depth; (iii) flow conditions such as water volume, flow timing and flow extremes; and (iv) energy sources characterized by the type, size and seasonal patterns of organic matter entering the stream. The biological component or 'biotic interactions' such as disease or parasitism was added as a fifth component some time later by Karr *et al.* (1986).

Physico-chemical water quality parameters were traditionally used as surrogates of other components of ecosystem integrity and, for example, water courses were classed as fishable or swimmable using only the physico-chemical condition of the ecosystem (Karr and Dudley 1981). However, under the definition of ecosystem integrity, the characterization of physico-chemical parameters is not enough to evaluate the integrity of freshwater ecosystems. For example, for this reason, over the last 30 years, there has been a trend towards the adoption of biological indicators to assess river health (e.g. Karr 1981; Wright 1995). The use of biological indicators has become popular for their capacity to integrate measures of the different components of ecosystem integrity in space and time. For example, Karr *et al.* (1986) argued that freshwater fish assemblages are especially recommended as biological indicators, because they respond to physico-chemical and biological perturbations and can provide integrated assessments over different spatial-temporal scales. Furthermore, the biological component of ecosystem integrity can only be assessed by the use of biological indicators. While there are instrumental methods to monitor water quality or evaluate changes in habitat structure, biological interactions can only be effectively quantified by using the organisms involved.

The index of biotic integrity

Different approaches have been proposed for using biological indicators in the assessment of ecological integrity. Some of these methods rely on the use of indicator species (Meador and Carlisle 2007; Hermoso *et al.* 2009) or assemblages (Wright 1995), so whenever a particular taxon or group of taxa are present or absent, we can infer specific physico-chemical and biotic conditions. These taxa would be classed as sensitive and would disappear when water quality declines. Some variants use the relative abundance or biomass of these organisms instead of their presence/absence (e.g. Kennard *et al.* 2005). However, one of the most successful ways of using biological indicators is through the combination of biological information into indices of biological integrity (IBI). The first IBI was proposed by Karr (1981) and was a fish-based index developed for rivers in north-east United States and was conceived as a multimetric index. These indices combine into a final score partial evaluations obtained from a set of independent metrics, which indicates the overall ecological integrity of the ecosystem being evaluated. The original IBI was composed of twelve metrics grouped into three main categories: taxonomic richness determined as the total number of species, habitat and trophic guild composition expresses as the proportion of insectivors or the proportion of top carnivores, and individual health and abundance which could be determined by the total abundance of taxa or the proportion of individuals with diseases. Each metric was designed to portray partial evaluations of the different components of ecological integrity in the final IBI score. For example, the abundance of lithophile nester species needing coarse substrates to spawn would indicate whether or not the physical structure of the habitat has been altered, perhaps through siltation. Karr's IBI has been the template that most of *a posteriori* fish-based indices have mirrored even the 30 years since its development (e.g. Aparicio *et al.* 2011; Schmitter-Soto *et al.* 2011; Terra and Araújo 2011) and across different continents [Africa, Kleynhans (1999); South America, Bozzetti and Schulz (2004); Europe, Pont *et al.* (2007); Asia, Hu *et al.* (2007); and Oceania, Joy and Death (2004)]. An indication of the popularity and success of this schedule is the exponential increase in the number of citations that Karr (1981) has received and the number of manuscripts published under the subject 'biological integrity' over this 30-year period (Fig. 1). The success of this approach to using biological indicators has led IBIs to be one of the most recognized and accepted tools to evaluate the ecological integrity of freshwater ecosystems elsewhere, and its use is frequent in international legislation.

Biological invasions, an overlooked threat in biological assessment

Despite the great advances in the maintenance and recovery of the ecological integrity of freshwater ecosystems resulting from the implementation of bioassessment programmes and the use of biological indicators, there are still problems that need to be overcome. Not all the components of ecological integrity have been equally treated, and biological issues are often wrongly evaluated or not even considered.

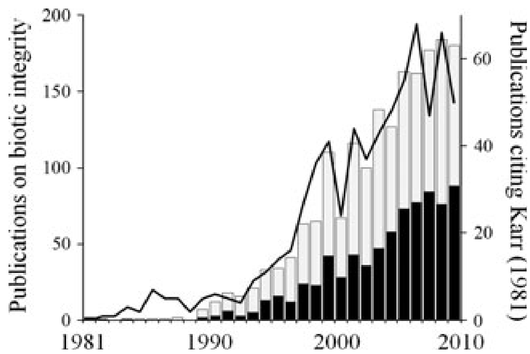

Figure 1 Temporal evolution (1981–2010) of the implantation of biotic integrity approaches in the scientific literature. Bars (left *Y*-axis) show the total number of published works, with their black portion reflecting the number of them that mentioned 'fish' in the title, abstract or keywords. The original search used 'biotic integrity' or 'ecological integrity' or 'biological integrity'. The line (right *Y*-axis) shows the yearly number of citations of Karr (1981). Searches were made using Thompson Scientifics' Web of Knowledge.

IBIs have traditionally overlooked the ecological consequences of biological invasions. We reviewed 83 IBIs developed in the period 1981–2011 across the whole planet, ruling out manuscripts that applied a previously developed and published IBI (Supporting information). We found that 74.7% of the reviewed IBIs did not mention the role of non-native species as a source of decline in ecological integrity. Furthermore, the sensitivity of the different IBIs to biological perturbations was not tested in a striking 96.4% of cases. The response of indices is usually validated against physico-chemical perturbations (e.g. Bozzetti and Schulz 2004; Ferreira *et al.* 2007b; Hu *et al.* 2007). In spite of this, testing the response of IBIs to biological perturbations associated with the presence of non-native species is necessary. This ensures that the whole range of features defining ecological integrity is being assessed. If this is not done there is a high risk of labelling ecosystems as having good ecological integrity with pristine physico-chemical conditions but in which non-native species dominate the fish assemblage. Furthermore, the incorporation of non-native species in conjunction with native species in the metrics is a common practice. 70.9% of reviewed IBIs included at least one metric such as the total number of species or the proportion of insectivorous individuals, based on data of the entire fish assemblage, irrespective of the species' origin. Non-native species have been shown to be sensitive to physico-chemical degradation (Kennard *et al.* 2005; Ferreira *et al.* 2007a). However, the combination of non-native and native species in the same metric constrains the capacity to detect the negative biological consequences that non-native species might be causing. For example, if non-native species drive the loss or decline of native assemblages, the use of combined metrics would not help to detect the underlying substitution process that might be happening in which native species were being replaced by non-natives. To demonstrate the effects of including non-native species in the calculation of metrics for the IBI, we give a hypothetical example in Fig. 2, where three native species have been substituted by three non-natives from a total of 11 species in a third-order river reach. This is not an unlikely scenario, given the high introduction and spread rate of non-native species into new catchments. For example, Leprieur *et al.* (2008) reported the presence of over 20 new non-native species in many catchments around the planet, and in some catchments, there was a higher number of non-native than native species (e.g. Olden and Poff 2005; Clavero and Garcia-Berthou 2006). The substitution of native species by non-natives would not affect some of the metrics in the IBI regardless of the trophic guild of the new species. This would lead to the overestimation of the true ecological integrity, since although the total number of species has remained the same retaining the high score for this metric, the integrity of the native assemblage has been undermined. In a more appropriate consideration of ecosystem integrity for this particular hypothetical example, this reach should receive a lower score (three instead of five in this example; Fig. 2) according to the decline in native species richness. This applies not only to commonly used metrics of taxonomic richness such as the total number of species, which is used in more than 90% of IBIs, or total abundance, but also to metrics of trophic guild, such as the proportion of top carnivores or insectivores. As a consequence underlying invasion processes as mentioned above can go unnoticed (Table 1).

The evaluation of ecological integrity is usually carried out by following the reference condition approach (Reynoldson *et al.* 1997). This relies on

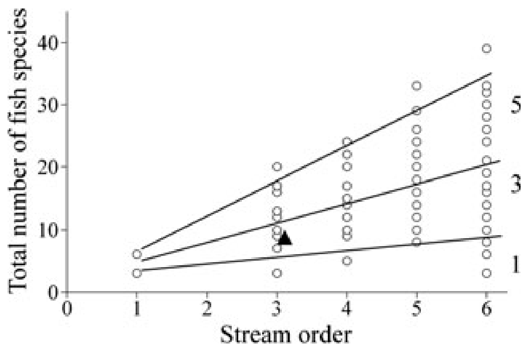

Figure 2 Simulation of the effect of including non-native species as part of one of the metrics included in Index of Biotic Integrity proposed in Karr *et al.* (1986) and showed in Table 1 ('total number of species' found in a sample). The variation in species richness along different stream order reaches in the Embarras River (Illinois, USA) is showed here. The lines represent the trisection used to score the metric while accounting for natural sources of variation (stream order in this case). A site in a third-order reach with 11 species would get the maximum score, for example, independently if these are native or non-native. If a biological degradation process occurred and three native species were replaced by three non-native species, the total number of species would remain constant, and the score would remain constant and insensitive to the biological change. An alternative metric that responds to this change would ideally incorporate only native species as showed with the black triangle. In this example, the site would get a score of 3 instead of 5 indicating that some sort of degradation has occurred given that the total number of native species has declined in relation to the expected in absence of perturbations.

the comparison of the observed condition of the ecosystem under evaluation with the expected condition in the absence of major perturbations, taken as a reference condition. The ability of bioassessment programmes to objectively evaluate ecological integrity depends largely on how well we can define these reference conditions (Hermoso and Linke 2012). Biological assemblages are known to change along environmental gradients (Rahel and Hubert 1991), so reference conditions should ideally account for these changes. Otherwise the evaluation obtained from the comparison will be prone to under- or overestimation errors. For example, if the ecological integrity of a low reach is evaluated using headwater fish assemblages as a reference condition, the evaluation will tend to overestimate the true integrity even when some perturbation might have taken place, species richness and diversity could be higher than expected for a headwater

Table 1 Example of insensitivity of Index of Biotic Integrity (IBI) to an underling invasion process. The value for each of the 12 original metrics in Karr *et al.* (1986) IBI for a third-order river reach in two different moments is shown. At t_1, the assemblage was in reference conditions, with 14 native species. At t_2, because of an invasion process, three new non-native species (predators) have appeared (it can be seen in the proportion of top carnivores), while three native species have disappeared because of predation pressure from the new predators (reduction in the number of darter, sunfish and sucker species), as well as the abundance of the remaining natives (proportion of sunfish and omnivores and total number of individuals). Note that as there has not been physico-chemical degradation, the number of intolerant species has remained constant. Despite the clear biological degradation from t_1 to t_2, neither the total IBI score nor each of the partial evaluations in each metric has changed. See Karr *et al.* (1986) for details on the scoring used for each metric.

	Metric	t_1	t_2	Score t_1	Score t_2
1	Total number of species	14	14	5	5
2	Number of darter species	4	3	5	5
3	Number of sunfish species	3	2	5	5
4	Number of sucker species	3	2	5	5
5	Number of intolerant species	4	4	5	5
6	Proportion of individuals as green sunfish (%)	5	1	5	5
7	Proportion of individuals as omnivores (%)	20	5	5	5
8	Proportion of individuals as insectivorous cyprinids (%)	45	20	5	5
9	Proportion of individuals as top carnivores (%)	5	50	5	5
10	Number of individuals in sample	50	20	5	5
11	Proportion of individuals as hybrids (%)	0	0	5	5
12	Proportion of individual with disease, or other anomalies (%)	0	0	5	5
	Total IBI score			60	60

reach. Karr *et al.* (1986) used a maximum-richness line approach to incorporate natural changes in the definition of reference conditions along streams of different order (see Fig. 2). Other commonly used methods to establish reference conditions are predictive models and classification methods. Predictive models estimate the expected assemblage composition for a given site according to habitat characteristics, while classification-based methods start identifying homogeneous classes using either environmental data leading to top-down classifications, or biological data, which results in bottom-up classifications. With a classification in hand, it becomes possible to establish common reference conditions for each class. Despite the importance of an accurate definition of reference conditions, 38.6% of the IBIs reviewed here did not account for natural changes in freshwater fish assemblages in their assessments. Furthermore, 88.5% of the IBIs established reference conditions paying attention only to physico-chemical degradation criteria. In this case, a site was not considered to be in a condition suitable for reference if some of the chemical parameters indicated that a perturbation had occured. It is commonly believed that non-native fish species mainly thrive in degraded environments, but they can also colonize sites in good physico-chemical condition (Kleynhans 2007; Hermoso *et al.* 2011). Therefore, defining reference conditions according to physico-chemical attributes alone hinders our capacity to correctly evaluate the effect of non-native species. If a site is used as reference but where the native assemblage has suffered a decline because of the invasion of non-native species, the evaluation obtained will tend to overestimate the true ecological integrity of new sites under evaluation (owing to underestimation of reference conditions). Ideally, only sites free of non-native species and in good physico-chemical condition should be used to define reference biological conditions. However, the spread of non-native species often makes it very difficult to find sites that are not invaded, so some authors have used thresholds on the relative abundance of non-native species to consider a site in reference condition (Kennard *et al.* 2006; Hermoso *et al.* 2010). This threshold aims to represent the limit where the abundance of non-native species is expected to have no or very low effects on native assemblage composition and abundance.

Biological invasions have not only been left out of IBI evaluations but also from the guidelines established by some legislation. For example, the Water Framework Directive (WFD, European Commission

2000) does not include non-native species as a potential source of degradation of ecological integrity and also does not encourage the application of methods to account for the effect of non-native species in the evaluation process. This has led to the development of IBIs that overlook the effect of non-native species across Europe (e.g. Pont *et al.* 2006, 2007; Ferreira *et al.* 2007b; Schmutz *et al.* 2007). Furthermore, the dearth of methods to objectively incorporate the degradation of the biological component of integrity in the evaluations also implies a complete lack of response to the increasing negative effects of non-native species. This is not a trivial issue, given the magnitude of invasions seen in many of the world's freshwater ecosystems (Leprieur *et al.* 2008) and the pernicious effects of non-native species on native assemblages (e.g. Olden *et al.* 2004; Hermoso *et al.* 2011). Although it might seem a paradox, the improvement of ecological integrity of European rivers and the fulfilment of the WFD exigencies could be indirectly achieved by letting non-native species become more widespread if the assessment methods remain insensitive to this threat. Some metrics would improve because of the new species entering a given ecosystem, such as intolerant or predatory species. Meanwhile the underlying process of degradation caused by the negative effect the new species has on the ecosystem would keep undermining the native assemblage.

Where to from here?

The biological degradation caused by the introduction of non-native species can seriously undermine the ecological integrity of freshwater ecosystems. However, current bioassessment approaches most often do not account for this sort of degradation. Can an ecosystem that is dominated by non-native species continue to be considered as having good ecological integrity? The current implementation of the concept of ecological integrity misses part of the opportunity that the use of biological indicators entails and it seems that we were using them to test something we could do more easily by using simple physico-chemical measures. As most commonly used, IBIs evaluate whether physico-chemical perturbations have surpassed the ecosystem's resistance and or resilience capacity and have thus affected biological assemblages. However, we still miss some of the features that define ecological integrity, such as the condition of the biological component of the system. This is of special concern as this component of ecological integrity cannot be evaluated by traditional methods. Although some of the metrics currently used do respond to changes in biological assemblages such as a decrease or an increase in species richness, they do not always distinguish between the native and non-native origin of the assemblage's elements. As we have discussed, this can constraint our capacity to evaluate the whole range of biological degradation and undermine our ability to respond to the threat associated with non-native species. We believe that the presence of non-native species, independently of the effect they may cause in the ecosystem, is a strong enough reason to reconsider labelling a system as having good ecological integrity, regardless of the physico-chemical conditions. We acknowledge that once a non-native species has established in a new ecosystem, it is very difficult and expensive to eradicate, so achieving high standards of integrity such as those aimed at by the WFD might be unrealistic as only sites with no non-native species could be labelled as in excellent condition. However, by overlooking the effect that this source of degradation has on native biodiversity and ecological integrity, we constrain our capacity to respond to one of this century's conservation challenges. Some major modifications are needed to change the situation. First, we need water management legislation to explicitly recognize the role of non-native species in the decline of native assemblages and the loss of ecological integrity. Second, in response to the latter we need to develop new methods to evaluate the ecological integrity of freshwater ecosystems that are sensitive to all sources of environmental degradation. These new tools must necessarily consider responses to physico-chemical derived changes in freshwater assemblages, but also to changes related to the existence of non-native species. To make these tools more sensitive to the latter and following our review, we would strongly recommend the following: (i) a more accurate definition of reference conditions, which explicitly account for the potential degradation caused by non-native species (see Kennard *et al.* 2005; Hermoso *et al.* 2010 for some examples) and natural gradients; (ii) avoid incorporating non-native species in IBIs (Hermoso *et al.* 2010) or, alternatively, include non-native species in separate metrics (Aparicio *et al.* 2011); and (iii) finally, to ensure that the IBI is suitable for evaluating the whole range of components of ecological integrity, we recommend explicitly evaluating its response to the presence and dominance of non-native species (Kleynhans 1999; An *et al.*, 1999; Hermoso *et al.* 2010). Only when this has occurred, we will have the bureaucratic support and appropriate tools to better tackle the difficult task of conserving biodiversity of freshwater ecosystems and evaluating and improving their ecological integrity.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Data S1. List of Indices of Biotic Integrity reviewed.

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