Water quality indices based on bioassessment: The biotic indices

Tasneem Abbasi and S. A. Abbasi

ABSTRACT

Water quality indices (WQIs), which translate numerical values of several water quality characteristics of a sample into a single value, play a very important role in the monitoring, comparison and control of water quality. The WQIs of modern and post-modern times have been almost exclusively based on physical and chemical characteristics, and have seldom included 'biological' characteristics other than biochemical oxygen demand (BOD) and faecal coliforms. During the last three decades of the 20th century, indices based on bioassessment have been increasingly used in some developed countries, besides South Africa and Serbia, to complement the conventional WQIs, but the rest of the world continues to base its WQIs predominantly on physical and chemical characteristics. The serious drawbacks of this approach have been elucidated in this paper and, against that background, the state-of-the-art of biotic indices has been summarized. The paper makes a strong case for greater reliance on bioassessment-based WQIs, especially by developing countries, to strengthen the diagnostic value of the conventional WQIs.

Key words | bioassessment, bioindicators, indices, macroinvertebrates, organic pollution, water quality

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INTRODUCTION

The water quality indices (WQI) of modern and post-modern times have been predominantly based on the assessment of physical and chemical characteristics, most often pH, dissolved oxygen, temperature, turbidity, hardness, total solids, nitrogen, phosphorus, some metals and some pesticides. Among 'biological' characteristics, only faecal coliforms and biochemical oxygen demand (BOD) have featured in most of these indices. Beginning from Horton's Index (Horton 1965), which is regarded as the first modern WQI, to the post-modern indices (Khan *et al.* 2003; Parinet *et al.* 2004; Sarkar & Abbasi 2006; Kannel *et al.* 2007), this has been a common feature of the frequently used indices. National water quality indices, such as the Canadian WQI (CCME 2001) and that developed by India's Central Pollution Control Board (Sarkar & Abbasi 2006) as well as known provincial or state indices such as the Oregon WQI (Cude 2001, 2002) and the British Columbia WQI (Khan *et al.* 2003), also follow this trend. One of the rare

exceptions is the Florida Stream WQI (SAFE 1995), which includes microinvertebrates in its repertoire.

In recent years increasing concern has been expressed for this near-total reliance on indices based on physicochemical parameters, to the neglect of biological parameters. There are two reasons for this concern.

First, any interpretation of water quality by physicochemical parameters is restricted to the parameters actually measured. For example we may find a water sample with its pH, salinity, hardness, BOD, chemical oxygen demand (COD) and so on, all within limits for drinking. But that water may contain harmful levels of some heavy metal, or some pesticide or even radioactivity! There are so many natural and anthropogenic chemicals that can be present in a water that it is practically impossible to analyse each and every one of the chemicals. The great ability of water to dissolve other chemicals – hence the term 'universal solvent' – adds to the difficulty in analysing any water

sample to its full extent. On the other hand aquatic organisms, especially the *community structure* of organisms such as plankton, macroinvertebrates, fishes and benthos, fairly reflect not only the current water quality but also the overall ecosystem health of a water body (Figure 1). It can even give an indication of the direction from which the ecosystem health has come to the point of analysis and where it is likely to go. Moreover, the state of a water body also reflects the state of the environment *around* it (Figure 2). This enhances the value of bioassessment even more. In other words, the community structure of the organisms (biota) of an aquatic habitat or an ecosystem integrates and reflects the cumulative effects of the factors impacting

that habitat or the ecosystem over time. Indices based predominantly on physicochemical parameters are unable to do so.

Second, it has been recognized that the under-utilization of bioassessment in the past with over-reliance on physicochemical assessment has been a major factor responsible for the deterioration of the ecological integrity of river ecosystems (Karr & Chu 1995, 2000; Roux *et al.* 1999a; Dallas 2002). Whereas North America, Australia, New Zealand, Central Europe and Western Europe have had biotic indices, other regions of the world, notably Asia, have been almost exclusively relying on 'abiotic' indices. This is a worrisome situation, made alarming in recent years because of
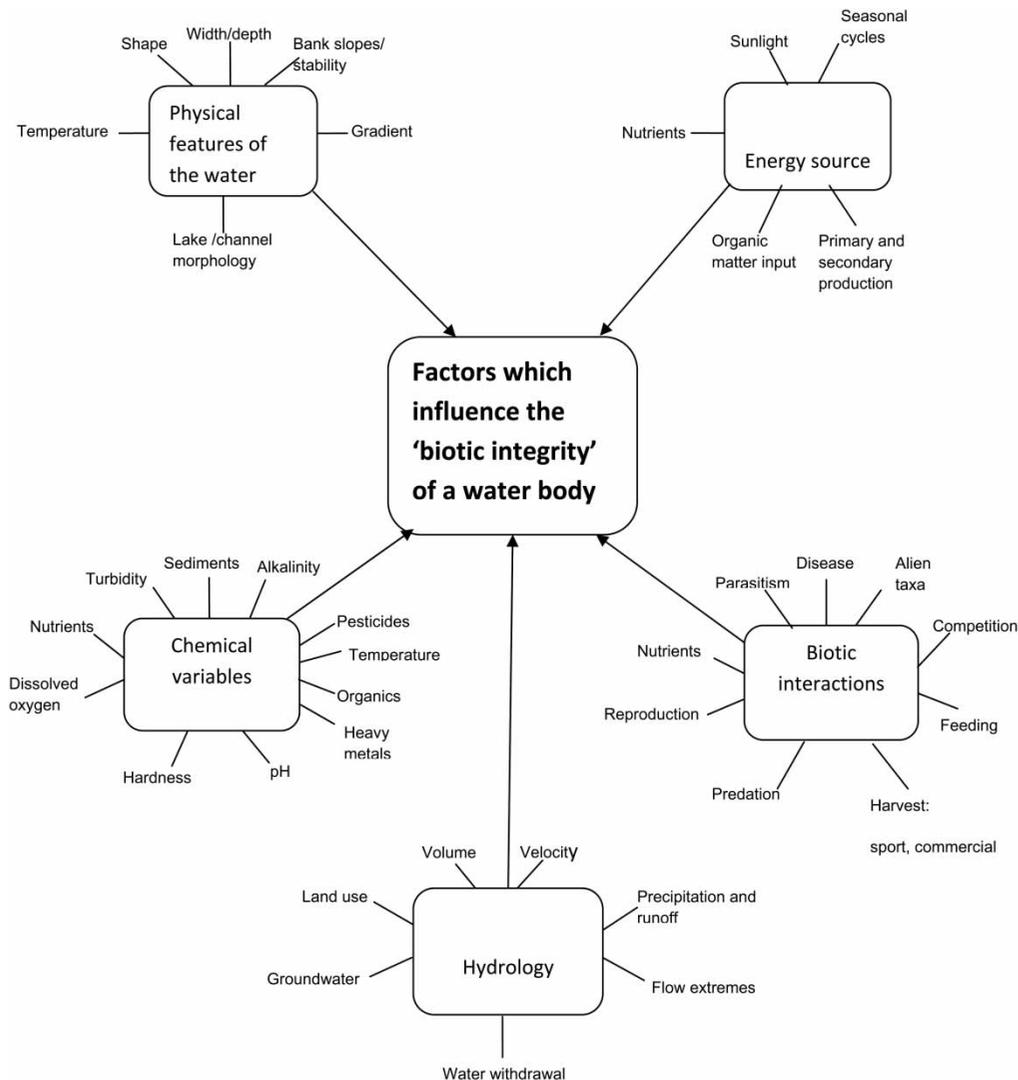

Figure 1 │ Factors which influence the biotic integrity of a water body.
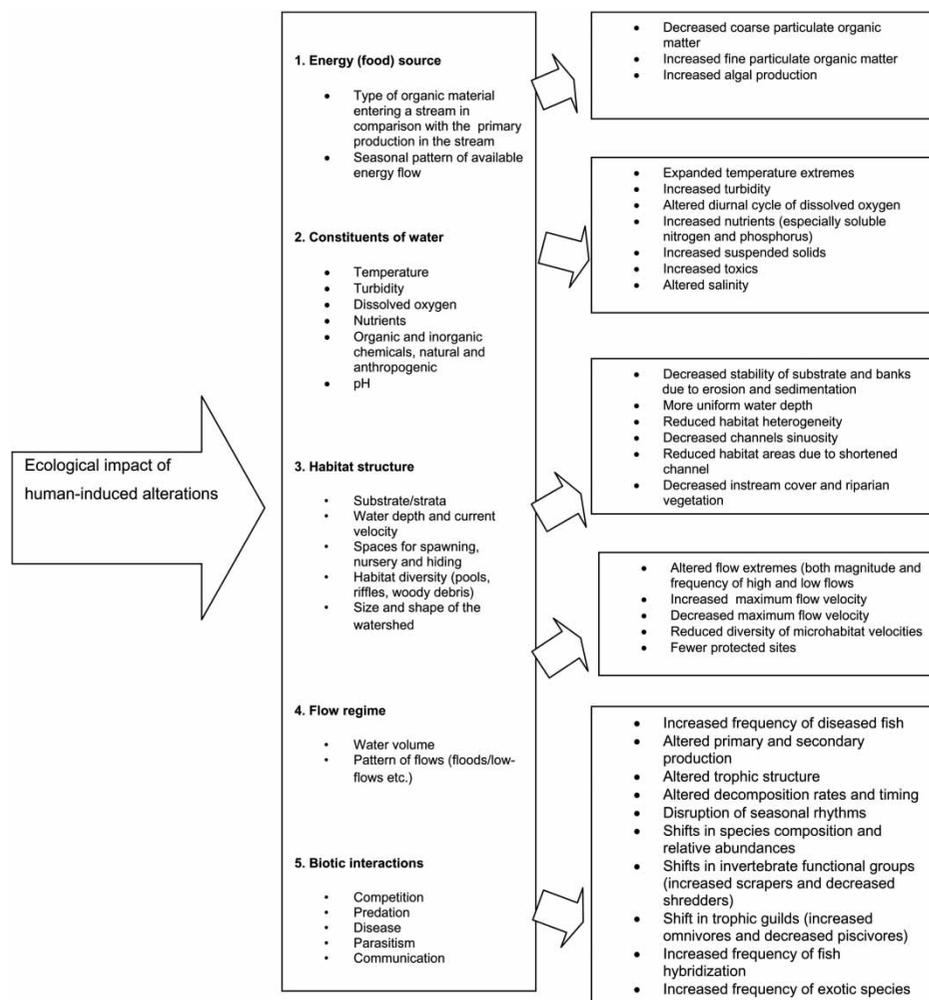
Figure 2 │ The myriad factors and cause–effect relationships which influence the biotic interactions of a water body and are influenced by it.

the hitherto unaccounted for, but likely to be very strong, *additional* impact of global warming on aquatic ecosystems (Abbasi & Abbasi 2010, 2011). There is an urgent need for much more widespread use of biotic indices in conjunction with the indices based on physical and chemical parameters.

Biotic indices in the context of the evolution of water quality indices

The concept of a water quality index (WQI) was introduced in its rudimentary form more than 160 years ago, when the presence or absence of certain organisms in a water source was used in Germany as an indicator of the fitness or otherwise of that water source. The development of that index came in the wake of the realization that human activities

produced pollution harmful to the biota (Davis & Simon 1995; Perry & Vanderklein 1996). Efforts were made to track the extent of biological degradation; the latter was even considered an indicator of the presence of human activities. The first-ever WQI was, thus, a 'biotic' index.

But, by and by, the focus of water quality evaluation shifted for much of the first half of the 20th century to the effects of chemical contaminants; rarely were connections between chemical criteria and ambient biotic condition documented. A few deviations to this general trend began to occur from 1964 onwards, but were largely restricted to the USA and some parts of Europe. Even now, as mentioned above, the use of biotic indices is very sparse, if used at all, in most of the developing countries. In 1964 the Trent Biotic Index (TBI) was developed for streams of Florida (USA).

In 1981 the first multimetric index – the Index of Biotic Integrity (IBI) – was introduced, also in the USA. The subsequent years have seen a slowly increasing reliance on biotic indices as a water quality management tool, especially in the developed countries. However, use of biotic indices is yet to catch on in developing countries. India is perhaps the most technologically advanced of the developing countries but there is little advancement here in this field and there is no accredited biotic index for water quality assessment.

The importance of even a single species in reflecting the water quality of a water source can be understood from the example of aquatic weeds (Gajalakshmi *et al.* 2001, 2002; Gajalakshmi & Abbasi 2004). If a pond or a lake is infested with weeds such as *Ipomoea* or water hyacinth, we can, with just one glance, say with certainty that the pond or the lake does not have clean water. If a lentic habitat is heavily choked with aquatic weeds we can also say, without any further experimentation, that it will be full of the larvae and pupae of mosquitoes and other insects, will have few edible fish, will have high BOD and COD, and so on.

Not all bioindicators are as obvious as aquatic weeds. Also not all water bodies are so grossly polluted that they are choked with aquatic weeds. But, as explained in the previous section, bioindicators *are* generally more 'expressive' than physicochemical parameters. Moreover, as elaborated in the next section, water quality assessment through physicochemical parameters represents a stressor-based monitoring approach while the same objective, when addressed through the monitoring of biota, represents the response-based monitoring approach. Both approaches have their distinguishing features and the ideal course is to use both in an integrated fashion. Consequently, biological indicators (or bioindicators) are increasingly becoming a key element of environmental and water resource management policies in most developed countries (Norris & Norris 1995; Moog & Chovanec 2000). Among the developing countries, only South Africa seems to have used biotic indices extensively, and Serbia to a lesser extent. India does not have any standardized or accredited biotic index and the authors have confined themselves to the use of general indices of species richness, diversity and evenness such as the Shannon Index and Pielou's Index (Ganasan & Hughes 1998; Chari & Abbasi 2003, 2004, 2005; Chari *et al.* 2003), which are not specific to water quality. Moreover, as

detailed below, these are not 'biotic indices' in the true sense of the term.

STRESSOR-BASED AND RESPONSE-BASED MONITORING APPROACHES

When the physicochemical quality of a watercourse is measured, it basically represents an attempt to see whether the water is clean or whether it carries one or more pollutant. In other words the monitoring of the water is done for possible 'stressors'. One may also be looking for biological stressors such as BOD or pathogens, but the known stressors are predominantly physical and chemical. The physicochemical indices are, therefore, stressor-oriented, a stressor being defined as any physical or chemical entity or process that can induce adverse effects on individuals, populations, communities or ecosystems (Thornton *et al.* 1994).

The stressor-oriented approach attempts, through stressor-specific quality criteria, to link stressors to possible biological responses. This predictive ability is, however, only possible where a known cause–effect relationship exists between a specific stressor and the biological component. But such cause–effect relationships, for a specific suite of conditions, can at best be explored with laboratory bioassays under controlled conditions, and may be far from applicable in real-life situations (Roux *et al.* 1999a). In the stressor-oriented approach the management focus is on the setting and use of rules for controlling the levels or concentrations of specific stressors, and this approach has a regulatory nature.

The other approach for environmental monitoring is 'response based', wherein the strategy is to assess the environmental health on the basis of the status of the responding organisms. It involves the monitoring of biological or ecological indicators in order to characterize the response of the environment to a disturbance. In turn 'disturbance' can be defined as any relatively discrete event in time that disrupts ecosystem, community or population structure and that changes the quality of natural resources, availability of substrata or the physical environment. The focus of response monitoring is on the effects resulting from the disturbance. It follows that the response-oriented approach indicates that something has or has not actually gone wrong in response to a stressor.

The collection and use of ecological data in the response-oriented approach is based on an ecosystem management and protection philosophy, in which the focus is on the status and behaviour of the environmental system being monitored and the status of that resource. Environmental response monitoring allows the measurement of how well an ecosystem is functioning, given the degree of perturbation to which it is subject.

From the comparison given in Table 1 it is clear that, for stressor and effects monitoring, the two underlying philosophies and the resulting assessments differ fundamentally. Both approaches have obvious uses and specific benefits in water quality management. The current thrust is towards operationally integrating the two approaches so that the resulting methodology incorporates the benefits of both.

Table 1 │ Characteristics of stress-oriented and response-oriented water quality monitoring approaches

The driver	Stress-oriented approach	Response-oriented approach
Monitoring focus	Stress causing environmental change; i.e. mainly chemical and physical inputs to aquatic systems	Effects (responses) resulting from natural and/or anthropogenic disturbances; e.g. changes in the structure and function of biological communities
Management focus	Water quality regulation: controlling stressors by regulating their sources (e.g. end-of-pipe focus)	Aquatic ecosystem protection: managing the ecological integrity of aquatic ecosystems (ecosystem or resources focus)
Measurement end points	Concentrations of chemical and physical water quality variables; e.g. pH, dissolved oxygen, copper	Structural and functional attributes of biological communities; e.g. diversity and abundance of benthic invertebrates
Assessment end points	Compliance or non-compliance with a set criterion or discharge standard	Degree of deviation from a benchmark or desired biological condition

An offshoot of the stressor-based monitoring approach is the concept of *carrying capacity* or *assimilative capacity* that gained wide currency during the 1990s. The concept was formulated around the use of the freshwater and marine environments for the disposal of mainly organic wastes and associated effluents. In this context, Cairns (1977) proposed that the assimilative capacity may be defined as the ability of an ecosystem to cope with certain concentrations of (organic) waste discharges, without suffering any significant deleterious biological effects.

But several assumptions are inherent in the utilization of the assimilative capacity concept in water-quality management:

a) Each environment has a finite capacity to accommodate some wastes without unacceptable consequences.
b) Such capacity can actually be quantified and subsequently utilized through allocation and management at acceptable impact levels.
b) Unacceptable consequences can be measured and quantified.
d) The utilization of the assimilative capacity will not have an injurious effect on those biological processes that contribute significantly to that capacity.
e) Zones of initial mixing, or *zones of allowed adverse ecological impact* may be required where significant ecological changes may occur.

Moreover, the assimilative capacity is generally very limited when dealing with toxic substances that are persistent and tend to concentrate in the environment and accumulate in aquatic biota (Abbasi 1976; Abbasi & Soni 1983, 1984). Also, the utilization of assimilative capacity must ensure a reference minimum flow condition that will minimize risk. Since cause-and-effect relationships in aquatic ecosystems are not well understood, it is not possible to accurately predict the degree of change that will result from a pollution input, especially when only chemical and physical constituents are being measured. Whereas it is possible to predict the assimilation of conservative constituents such as BOD with a fair degree of reliability, that of persistent and toxic substances is fraught with a large degree of uncertainty. The uncertainty is even greater where chemical interaction of multiple stressors, for example in a complex effluent, occurs. Therefore, the stressor-oriented monitoring and

management approach should be complemented with the response-oriented approaches (exemplified by the biotic indices) for better water quality management.

Figure 3 illustrates how biomonitoring techniques can be used to quantify biological condition over time. The segment A represents a natural range of variation – this range reflects the condition that will occur, without human interference, within a specific region; it also serves as 'control' or reference region. The segment B signifies a zone of 'acceptable' range of variation or change – the lower end of this zone will determine the lowest quality objective that can be set and managed for. This zone can be subdivided on the basis of more specific biological impact criteria. Political, economic and ethical issues may also be taken into consideration.

The segment C represents an unacceptable range of variation or change – it is essentially a cushion between the lowest allowable management objective and the point where the ecosystem loses its resilience (becomes irreversibly damaged). The cushion provides the safety margin necessary because of the uncertainties associated with ecological factors. The higher the uncertainty and hence unpredictability of what would constitute an irreversible change in a specific region, the bigger the safety margin that should be employed. An example of such ecological uncertainty is the degree to which biota will adapt to selection pressures when a system experiences a certain level of perturbation.
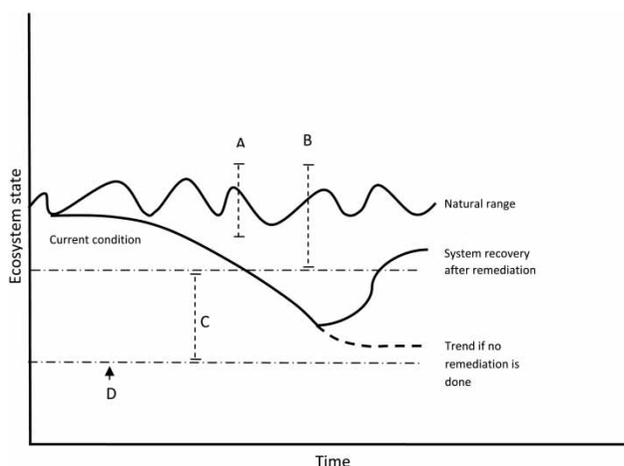

Figure 3 │ Naturally functioning ecosystem (A), acceptable range of fluctuations (B), perturbations beyond acceptable range but which can be reversed by remediation (C), and the critical threshold beyond which an ecosystem becomes irretrievable (adapted from Roux et al. 1999a).

The critical threshold is represented by the segment D. Beyond this threshold the ecosystem will be disturbed irreversibly; it will not be able to recover to its natural equilibrium state (A) or even to an acceptable or desired equilibrium state (B).

From Figure 3 it is clear that only with dependable data collected over the long term can a distinction be made between natural and unnatural ranges of variation in an ecological system. The fact that each ecosystem has its own, a unique, regime of natural variation adds to the complexity of this task. The manner in which unnatural variations occur also differs according to the type of impact. Furthermore, it may be difficult to identify and characterize variations where there is a collective forcing by anthropogenic and natural impacts.

BIOTIC INDICES: GENERAL

Biotic indices summarize and present as simple, numeric figures the biological community structure. As with the physicochemical quality-based indices mentioned above, the biotic indices also allow the results to be communicated in a way that is understandable to natural resource managers, decision-makers, politicians and the general public (Resh 1995; Uys et al. 1996; Stark 1998).

Three basic types of index can be generated (Johnson et al. 1993): diversity indices, comparison (similarity or dissimilarity) indices and biotic indices. The Shannon-Weaver index, the Simpson's diversity and dominance indices and Pielou's evenness index, among others, are well-known examples of the first two classes of indices. These indices have been used extensively for aquatic biota and even more extensively for terrestrial and avian biota (Abbasi & Vinithan 1999; Chari et al. 2003). But these indices overlook many important variables and tend to oversimplify the natural systems which are, in fact, highly complex (Karr 1981). For example the Shannon-Weaver index can merely say that two polluted regions are equally diverse but cannot say which of the two contains more beneficial or which one more harmful organisms. Indices have also been developed to assess the risk of pollution due to industrial accidents (Khan & Abbasi 1997a, b, 1998a, b, 1999a, b, 2000a, b, 2001); these indices include impacts on the quality

of water resources but none has been specifically applied to aquatic systems. Hence these indices are not further discussed here. In contrast, biotic indices are more 'expressive' and revealing of ecological health. In biotic indices, each taxon from a particular group of organisms is assigned a sensitivity weighting, or a 'score', based on the tolerance or sensitivity of that taxon to particular pollutants. The scores of all the individual taxa sampled at a site are summed and/or averaged to provide a value by which the ecological health of the biotic community, hence the health of the water body, can be gauged. Some biotic indices include abundance estimates in the scoring system.

The saprobien or saprobic system, which stems from the research work of Kolkwitz and Marsson on German rivers in the early 1900s, is generally considered to be the first biological scoring system for the assessment of water quality in river ecosystems (Washington 1984; Rico et al. 1992; Knoben et al. 1995; Verdonschot 2000; Sandin et al. 2001). Indices based on the saprobien system are determined by the presence and absence of specific indicator species from a number of different groups and trophic levels (mainly bacteria, algae, protozoans and rotifers, but including some benthic invertebrates and fish) for which the tolerances to organic pollution have been established (Metcalfe-Smith 1994). Selected components of the total aquatic community are thus used as an indicator for the degree of organic pollution (Friedrich et al. 1996). Most modern biotic indices, on the other hand, are based on the presence and pollution-tolerances of the community of organisms sampled from a particular group, such as the benthic macroinvertebrates (Ollis et al. 2006).

In recent years, to optimize the use of the time and resources available for ecological assessments, there has been increasing emphasis on the use of biotic indices based on community-level rapid bioassessment techniques (Brown 2001; Dallas 2002; Metzeling et al. 2003). The latter, which usually involve qualitative (or semi-quantitative) sampling with few or no replicates and limited taxonomic resolution, have been developed to inexpensively monitor problem areas and thereby provide inputs for decision-makers as to where more intensive and quantitative studies for arriving at corrective steps need to be undertaken (Resh et al. 1995; Ollis et al. 2006). Numerical simulations to assess the sensitivity of the values of two biotic indices to the

sample size and taxonomic resolution by Bigler et al. (2010) reveal that instead of the stipulated count of 400, a count of just 40 diatom values for 50 streams, and 80 values for 60 streams, were sufficient to obtain the same index classification. Further, excluding rare taxa had negligible effect on the indices. These results indicate that it may be possible to adopt reduced taxonomical resolution for some biotic indices for improving the economics of stream monitoring, without sacrificing precision. But this conclusion is specific to the diatom-based indices studied by the authors and cannot be generalized. It can be said that rapid assessment techniques are not a replacement for more traditional quantitative studies and detailed biological surveys, but rather a precursor to these.

Attempts have also been made to employ machine learning (artificial intelligence) techniques such as artificial neural network and genetic programming for selecting ecologically significant input variables in environmental prediction (Chau et al. 2002; Zhao et al. 2006; Muttil & Chau 2006, 2007; Wu & Chau 2006).

One of the most critical issues in any bioassessment is the identification of reference ('control') sites and reference conditions. Such sites should be truly reflective of natural, unpolluted conditions and thus serve as reference or 'control' sites with which the test sites can be compared to know whether a certain impact causes an aquatic assemblage or ecosystem to respond in some way that is outside the natural range of variation (Roux et al. 1999b). In other words the ultimate objective of any bioassessment programme is to facilitate the detection of disturbance at a site, as reflected by one or more components of the biota. Reference conditions facilitate this by defining what is expected at a site and provide a means of comparing observed conditions with expected conditions so that the degree of impairment or deviation from natural conditions can be determined.

Unfortunately, due to widespread human encroachments everywhere, it is very difficult to find non-impacted sites in most regions, especially in lowland areas, for use as reference or 'control' sites. Consequently, minimally disturbed or least-impacted of the available sites are generally used to determine the *best attainable* reference condition (Roux & Everett 1994; Reynoldson et al. 1997; Norris & Thoms 1999; Verdonschot 2000). Once the best attainable

reference conditions have been established for the aquatic ecosystems of a region, these can be used as benchmarks to classify the degree of impairment at monitoring sites (Gerritsen et al. 2000; Dallas 2002) and can form a scientific basis for setting ecological resource quality objectives (Roux & Everett 1994; Roux et al. 1999a).

In an attempt to circumvent the problem of the near-absence of pristine reference sites in the contemporary world, Lavoie & Campeau (2010) have developed an innovative method for assessing past conditions of streams on the basis of diatom assemblages extracted from the guts of fish stored in museums. By using the Canadian diatom index they were able to compare stream conditions for the 2003–2007 periods with the conditions prevailing in 1925–1948. More work along these lines may be helpful in solving the problem of setting appropriate frames of reference for the present-day monitoring.

An estimate made by the EPA office of Ohio State, USA (Yoder 1989), gives an indication (Table 2) of the relative cost of physicochemical assessment of water quality in comparison with bioassays and bioassessments (biotic indices). It is at best an illustrative, region-specific assessment, yet it does reveal that the cost of biotic index-based water quality

Table 2 | Comparative costs of physicochemical analysis, bioassays and index-based bioassessment of the quality of a water resource (Yoder 1989)

Domain	Per sample* (US$)	Per evaluation (US$)
Physicochemical water quality		
4 samples/site	1,436	8,616
6 samples/site	2,154	12,924
Bioassay		
Screening (acute 48-h exposure)	1,191	3,573
Definitive (LC50* and EC50 48 and 96 h)	1,848	5,544
7-d (acute and chronic effects 7-d exposure single sample)	3,052	9,156
7-d (as above but with composite sample collected daily)	6,106	18,318
Index-based bioassessment		
Macroinvertebrate community	824	4,120
Fish community	740	3,700
Fish and macroinvertebrates (combined)	1,564	7,820

*1989 values.

assessments is the least of the three options. It must be emphasized, however, that water for physicochemical parameters can be sampled and analysed more quickly than the conducting of biological sampling and identification. In addition, instrument-based continuous monitoring of physicochemical quality is possible with real-time transmission of data. Physicochemical quality can also be assessed, to some extent, by remote sensing. All these advantages are not available with bioassessment of aquatic organisms. Moreover, only physicochemical analysis can identify specific pollutants that may be stressing the biota of a water body.

ORGANISMS COMMONLY USED IN BIOASSESSMENT

Various organisms have been used in the bioassessment of the water quality and ecological integrity of aquatic ecosystems, including bacteria, protozoans, diatoms, algae, macrophytes, macroinvertebrates and fish (Dallas & Day 1993, 2004; Barbour et al. 1999; Milner & Oswood 2000; Brown 2001; Meloni et al. 2003; Zgrundo & Bogaczewicz-Adamczak 2004; Suárez et al. 2005; Moreno et al. 2006; Lavoie et al. 2008; Maggioni et al. 2009). Of these, benthic macroinvertebrates are the most widely used group (Resh et al. 1995; Dallas 2002), especially for lotic systems (Moog & Chovanec 2000; Sandin et al. 2001; Fabela et al. 2001).

BIOTIC INDICES BASED ON MACROINVERTEBRATES

There are several advantages in using benthic macroinvertebrates in bioassessment (Ollis et al. 2006; Mugnai et al. 2008; Lavoie & Campeau 2010). Benthic macroinvertebrates are largely non-mobile, ubiquitous and relatively abundant inhabitants of both lotic and lentic habitats. There are often many species within a community with varying sensitivities to stresses and relatively quick reaction times, resulting in a spectrum of graded, recognizable responses to environmental perturbation. Also responses to different types of pollution have been established for many common species. Macroinvertebrates have life cycles that

are long enough for temporal changes caused by perturbations to be detected, but short enough to enable the observation of recolonization patterns following perturbation. They are relatively easy and inexpensive to collect, particularly if qualitative sampling is undertaken, and are well suited to the experiments required for biomonitoring.

In addition to the advantages associated with sampling macroinvertebrates, methods of analysing their data are also well established. Consequently, numerous biotic indices have been developed for the assessment of river ecosystems that are based on aquatic macroinvertebrates. Brief descriptions of the more important or widely used indices, listed in chronological order, are provided below.

Beck's Biotic Index (Beck's BI), 1954

Beck is perhaps the person who coined the term 'biotic index' (Washington 1984); he surely is the one who popularized it (Davis 1995). Beck's BI, developed for streams in Florida, is considered to be the first true biotic index; this index is based on the relative tolerances of macroinvertebrates to organic pollution, with field-sorting undertaken and identification to species level. Species known to be intolerant to slight organic pollution ('Class I organisms') and those known to be tolerant of moderate organic pollution ('Class II organisms') are distinguished from the rest of a sample. The final index value for a site is calculated by summing the number of species of Class I organisms, multiplied by two, and the number of species of Class II organisms. A single value ranging between 0 and approximately 40 is generated, with values greater than 10 indicating unpolluted sites and values between 1 and 6 indicating moderately polluted sites.

Trent Biotic Index (TBI), 1964

The Trent Biotic Index (TBI) (Woodiwiss 1964), on which several other modern biotic indices are based (Indice Biotique, Chandler's Biotic Score, Chutter's Biotic Index, Hilsenhoff's Biotic Index, Biological Monitoring Working Party, Belgian Biotic Index and Danish Stream Fauna Index), was developed by the Trent River Authority in England. Qualitative, combined sampling of all available habitats is undertaken for 10 minutes by means of a hand-net. A single value is generated by the index, ranging from 0 (grossly polluted) to 10 (unpolluted). The value at a site is determined by the presence or absence of six key types of invertebrate with varying degrees of tolerance to organic pollution, together with the number of specific 'groups' identified to family, genus or species levels.

Indice Biotique (IB), 1968

The Indice Biotique (IB) (Tuffery & Verneaux 1968) was derived from the TBI, for use in France. Lotic and lentic habitats are sampled separately using Surber and grab samplers, respectively, and two indices are calculated: a lotic sub-index and a lentic sub-index. Index values for the IB are determined by the presence of key groups and the number of predefined taxa (or 'systematic units', identified to family, genus and species levels) in each sample, with laboratory-based identification. The IB was modified into the Indice Biologique de Qualité Générale (IBQG), which introduced a greater number of indicator groups and the sampling of eight different habitats at a site, defined on the basis of substrate and velocity conditions. The Indice Biologique Global (IBG), which is based on the IBQG, was adopted as the standard bioassessment method throughout France (Metcalfe-Smith 1994). The IBG was superseded by an updated version known as the Indice Biologique Global Normalisé (IBGN). With the IBGN, lotic habitats are sampled with a Surber sampler and lentic habitats with a hand-net (both 500 µm mesh). The modifications differ from the original IB in that faunal groups are mostly identified to family level.

Chandler's Biotic Score (CBS), 1970

Chandler's Biotic Score (CBS) (Chandler 1970), originally developed for upland rivers in the Lothians region of Scotland, is based on the TBI. However, unlike the TBI, it includes an abundance factor in the final calculation of the index score and only riffle (stones-in-current) areas are sampled with a hand-net (1,000 µm mesh size) for a total of 5 min. The total score is determined by summing the pollution tolerance scores for each defined 'group' of invertebrates sampled (identified to genus or species), with a sliding scale for individual scores based on the estimated

level of abundance. There is no upper limit for the final CBS value, but unpolluted sites generally have scores greater than 3,000 (Johnson *et al*. 1993).

The Average Chandler Biotic Score (Avg. CBS), a modification of the CBS system with the final score for the number of groups present in a sample normalized, was developed because the original system generated low scores for unpolluted, headwater sites (Murphy 1978, Johnson *et al*. 1993). This normalized scoring system, which generates values ranging from 0 (severely polluted) to 100 (unpolluted), is more reliable than the original CBS system at discriminating between polluted and unpolluted sites (Washington 1984) and has been found to be a relatively robust indicator of water quality.

Chutter's Biotic Index (CBI), 1972

This system, which is loosely based on the TBI (Metcalfe-Smith 1994), involves sampling the stones-in-current habitat with a hand-net or Surber sampler (mesh size 290 μm). A spectrum of 'quality values' has been determined for an extensive list of predefined taxa (identified to various taxonomic levels), based on the known occurrence of the defined groups in polluted waters. The final CBI value, which ranges from 0 (unpolluted) to 10 (severely polluted) and represents the average quality value for the organisms sampled, is calculated by dividing the sum of the individual scores for all the taxa sampled by the total number of individuals in the sample. The CBI was never widely used because it requires advanced taxonomic skills, and is time-consuming and expensive to apply (Chutter 1994, 1995, 1998).

Hilsenhoff's Biotic Index (HBI)

Hilsenhoff's Biotic Index (HBI) (Hilsenhoff 1987) is an adaptation of the CBI, and was developed for evaluating organic and nutrient pollution in streams in the Wisconsin region of North America.

The original HBI has been refined in recent years by limiting the number of individuals scored in each taxon to 10, which remedies some problems commonly encountered with the system and reduces seasonal variability in the index value (Hilsenhoff 1998). The HBI, with tolerance values modified for specific geographic regions, is regularly used for water quality assessments in many states across North America (Reynoldson & Metcalfe-Smith 1992).

Biological Monitoring Working Party (BMWP) Score System

In this system, which was introduced in 1978 and modified in 1980 and 1983, all major aquatic habitat types are sampled with a pond-net of 90 μm mesh-size for a total of 3 min and taxa are identified in the field. The score values for all the predefined invertebrate families present in the sample for a site are summed to give the total BMWP score. It is divided by the number of taxa sampled to determine the average score per taxon (ASPT) for the site. The BMWP-ASPT index has proved to be a relatively robust measure of water quality for rivers in the United Kingdom (Pinder *et al*. 1987; Metcalfe-Smith 1994).

Belgian Biotic Index (BBI), 1983

The Belgian Biotic Index (BBI) (De Pauw & Vanhooren 1983) combines the sampling procedure of the TBI and the scoring system of the IB, but with lotic and lentic habitats scored together. All available habitats are sampled with a 300–500 μm mesh hand-net for a total of 3 min (for rivers less than 2 m wide) or 5 min (for larger rivers). Collected macroinvertebrates are preserved *in situ* and taken back to the laboratory for identification, mainly to family or genus levels. The final index value ranges from 0 (very heavily polluted) to 10 (unpolluted), with values less than 5 indicating that the situation is critical. The BBI has been successfully applied throughout Belgium and in other countries, including Spain, Algeria, Luxembourg, Portugal and Canada (Metcalfe 1989). It is currently used in Belgium and some surrounding countries (Metcalfe-Smith 1994; Iversen *et al*. 2000).

Macroinvertebrate Community Index (MCI), 1985

The Macroinvertebrate Community Index (MCI) (Stark 1985), developed for assessing water quality in New Zealand streams, is based on the BMWP method and is similar to the CBI and HBI. Scores are allocated to a list of predefined taxa based on their pollution tolerances, with values from

1 (extremely pollution tolerant) to 10 (extremely pollution sensitive). The final index value for a site is calculated by summing the tolerance values for each taxon present in a sample, dividing by the number of taxa sampled and multiplying by a scaling factor of 20. Although the MCI can theoretically range between 0 and 200, in practice it rarely exceeds 150, with scores greater than 120 indicating pristine conditions and scores less than 50 indicating extreme pollution (Stark 1993).

Iberian BMWP (IBMWP/BMWP), 1988

The Iberian BMWP (IBMWP) (Bonada 2003) is also an adaptation of the BMWP system. It is a qualitative or semi-quantitative method that uses a kick-net with 250 µm mesh size and field-based macroinvertebrate identification to family level. All available habitats are successively sampled over a 100 m stretch of river until no new taxa are recorded.

The final IBMWP score, number of taxa and IASPT (IBMWP score divided by number of taxa) are calculated for a site based on all the taxa collected and observed. Separate indices can also be calculated for lotic and lentic habitat groups, if they have been collected and analysed separately. Abundances are estimated according to the following ranks: 1 (1–3); 2 (4–10); 3 (11–100); 4 (>100) (Bonada 2003). Although these abundance estimates are not used to calculate the final indices, they aid in the interpretation of IBMWP results. The IBMWP has been shown to be effective for the bioassessment of the Spanish rivers and, in 1991, it was adopted by the Spanish Society of Limnology for use throughout the Iberian Peninsula (Zamora-Muñoz & Alba-Tercedor 1996).

Rivers of Vaud (RIVAUD) Index, 1989, 1995

The RIVAUD Index (Lang et al. 1989) was developed to assess the water quality of rivers in the canton of Vaud in western Switzerland. The method involves the collection of macroinvertebrates from the stones-in-current biotope, using kick-sampling techniques and a hand-net with a mesh size of 400 µm. Each sample consists of the macroinvertebrates collected from six areas of 0.1 m², with the combined list of taxa from one spring sample and one summer sample used to analyse a sampling site. Macroinvertebrates are identified to family and/or genus level. The final index value, which ranges from 0 to 10, is calculated by adding the allocated score for the number of taxa (grouped into six classes of values with allocated scores of 0–5) and that for the number of intolerant taxa (also grouped into six classes of values with allocated scores of 0–5). Intolerant taxa are taken to include Heptageniidae, Plecoptera and case-bearing Trichoptera. Non-hierarchical cluster analysis of the data collected over five years from 162 sampling sites along 51 rivers in western Switzerland was used to delimit classes of values for the total number of taxa and number of intolerant taxa (Lang et al. 1989). The system has been designed so that RIVAUD Index values of 0–3 indicate poor water quality, with values of 4–6 indicating average water quality and values of 7–10 indicating good water quality.

An updated version of the RIVAUD Index was developed by Lang & Reymond (1995), based on additional data collected mostly from the same rivers as those used to initially develop the RIVAUD Index. This updated version was called RIVAUD 95, after the year of its development. The sampling method for RIVAUD 95 is the same as that for the original index system, except that additional late summer samples are collected from rivers in the Alps to ensure that seasonally restricted taxa are captured in this region.

Stream Invertebrate Grade Number – Average Level (SIGNAL) Biotic Index, 1995

The SIGNAL Biotic Index (Chessman 1995, 2003) was initially developed in 1995 for the assessment of water quality in the Hawkesbury-Nepean River system of New South Wales, eastern Australia (Chessman 1995) and later modified in 2003 to broaden its applicability to the whole of Australia (Chessman 2003). Macroinvertebrates are collected from six predefined habitats present at a site. Riffles, pool edges and aquatic macrophytes are sampled with a hand-net (250 µm mesh), pool rocks and submerged wood are removed from the stream by hand, and soft sediment samples in deep lowland rivers are obtained with a grab sampler and then sieved through 250 mm mesh. The sampling time is not stipulated.

Instead, for each habitat type, 100 invertebrates in total are collected with no more than 10 specimens per taxon.

Specimens are preserved and taken back to the laboratory for identification to family level. Sensitivity grades ('SIGNAL 1 grades') ranging from 1 (pollution tolerant) to 10 (pollution sensitive) were initially assigned to widespread families of macroinvertebrates in river systems of south-eastern Australia (Chessman 1995). Modified 'SIGNAL 2 grades' were subsequently derived for macroinvertebrate families occurring across Australia (Chessman 2003).

Danish Stream Fauna Index (DSFI), 1998

The Danish Stream Fauna Index (DSFI) is based on the TBI, but both positively and negatively scoring diversity groups are used. Also, sampling involves kick-sampling of all available habitats along each of three transects, at four equidistant points across the width of the stream, with transects approximately 10 m apart (placed diagonally across the stream if stream width is less than 1 m). The 12 kick samples, which are obtained using a hand-net with 500 μm mesh size, are combined for further analysis, and 5 min of hand-picking from submerged stones and large wooden debris is carried out. The pooled kick sample and the hand-picked sample are preserved separately in the field, with identification (to genus and family level) undertaken in the laboratory, keeping the two groups of samples separate.

The final index value for the DSFI varies from 1 (severely impaired) to 7 (best ecological quality). It is calculated by taking into account the number of diversity groups (i.e. the number of positive groups of taxa minus the number of negative groups of taxa, based on a list of positive and negative taxon groups) and the presence of particular indicator groups of taxa in the total fauna sample (i.e. kick samples plus hand-picked sample from each site). The final DSFI index value is obtained from a matrix table that has four categories for the number of diversity groups as columns and six indicator groups (with corresponding lists of indicator taxa) as rows.

Balkan Biotic Index (BNBI), 1999

The Balkan Biotic Index (BNBI) (Simić & Simić 1999) was developed on tributaries of the Danube River in Serbia, for river water quality assessment in the Balkan Peninsula. Loosely based on the CBS, the BNBI requires an estimation of the abundance of sampled macroinvertebrates. It incorporates measures of the dominance and constancy of the taxa sampled, together with a measure of the diversity of the macroinvertebrate community at a sampling site. The BNBI ranges from 0 (for heavily polluted waters) to 5 (for very clean waters).

A rapid reckoner of the attributes and modus operandi of these indices is presented in Table 3.

BIOTIC INDICES AS INDICATORS OF WATER SAFETY AND HUMAN HEALTH RISKS

Biotic indices have been very commonly and successfully applied to the monitoring of major stressors such as BOD, COD and plant nutrients vis-à-vis concerns for water safety and human health. But there is strong evidence that water micropollutants (such as pesticides and heavy metals) can also dramatically alter the structure and physiology of benthic communities (Ivorra 2000; Blanco et al. 2007; Blanco & Bécares 2010), in turn having their toxic impact manifest in the scores of the indices concerned. For example, several significant statistical correlations between diatom indices and the water concentrations of heavy metals were noted by Sabater (2000). Zamora-Muñoz et al. (1995) observed significant correlations between BWMP and IBWMP and concentrations of Cu, Zn, pesticides, detergents, fats and oils. Among toxicological factors in rivers, concentrations of metals and other pollutants such as total PAHs (polycyclic aromatic hydrocarbons) have been seen to influence variance in some macroinvertebrate indices (Pinel-Alloul et al. 1996). Significant correlation of a biotic index with Cu and Pb was also noted by Robson et al. (2006).

In general, shifts from sensitive to tolerant taxa occurring in polluted environments are utilizable as indicators of metallic pollutants and toxic organics by biotic indices as effectively as the monitoring of agents of eutrophication by these indices (Robson et al. 2006). In other words indices based on invertebrates are well suited to assessing this kind of impact, given a little modification to improve their

Table 3 | An overview of the major biotic indices, based on aquatic macroinvertebrates

Biotic index	Biotypes sampled	Sampling equipment	Sampling protocol	Taxonomic level	Score range	Regions in which currently used
Beck's Biotic Index	All, combined	Not stipulated	Non-quantitative	Species	0–c.40	–
Trent Biotic Index	All, combined	Hand net	Non-quantitative	Family + genus + species	0–10	–
Indice Biotique	Lotic + lentic, separate	Surber + grab	Semi-quantitative	Family + genus + species	0–10	–
Chandler's Biotic Score	Stones-in-current	Hand net (1,000 μm)	Semi-quantitative	Genus + species	0–?	USA
Chutter's Biotic Index	Stones-in-current	Hand net/ Surber	Quantitative	Family + genus + species	0–10	–
Hilsenhoff's Biotic Index	Stones-in-current	Hand net	Quantitative > 100	Genus + species	0–10	USA
Biological Monitoring working party	All, combined	Hand net	Non-quantitative/ semi-quantitative	Family	0–c.200	UK, Finland, Sweden
Belgian Biotic Index	All, combined	Hand net	Non-quantitative	Family + genus	0–10	Belgium and surrounding countries
Macroinvertebrate Community Index	Stones-in-current	Hand net/ Surber	Non-quantitative	Genus	0–200	New Zealand
Iberian BMWP	Lotic + lentic, combined/ separate	Hand net	Non-quantitative	Family	0–c.200 0–10 (ASPT)	Spain, Italy
Rivers of Vaud Index, 1995 version	Stones-in-current	Hand net	Semi-quantitative	Family + genus	0–20	Western Switzerland
Stream Invertebrate Index	6 per-defined	Hand net	Non-quantitative, 100 organisms	Family	0–10	Australia
Danish Stream Fauna Index	All, combined	Hand net (500 μm)	Semi-quantitative, 12 samples	Family + genus	0–7	Denmark, Sweden
Balkan Biotic Index	All, combined	Benthos net	Quantitative	Family + sub-family + genus	0–5	Serbia

sensitivity and performance (García-Criado *et al.* 1999; Blanco & Bécares 2010).

BIOTIC INDICES AND DEVELOPING COUNTRIES

Biotic indices have been successfully used for the bioassessment of rivers in many parts of the world, mostly the developed countries and some developing ones. But a large number of developing countries have not yet started using biotic indices. For example, biotic indices have not to date been developed or used to any significant extent in Latin America (Pringle *et al.* 2000), Central and Eastern Asia (Li *et al.* 2000), or South-east Asia (Dudgeon *et al.* 2000). Among the exceptions are the use of indices based on benthic macroinvertebrate communities in Nicaragua (Fenoglio *et al.* 2002), Macedonia (Lazaridou-Dimitriadou *et al.* 2004), Vietnam (Duong *et al.* 2007) and Brazil (Mugnai *et al.* 2008). On the Indian subcontinent, no biotic indices are used for assessing the water quality of

rivers because none of the currently available biotic indices from other countries has been found to be entirely suitable (Gopal *et al.* 2000; Sarkar & Abbasi 2006).

LIMITATIONS OF BIOTIC INDICES

Despite their proven utility in rapid bioassessments, biotic indices must be carefully interpreted using supplementary data and their significant limitations must always be borne in mind. These include the restricted applicability to a particular geographic area and/or type of stressor (Washington 1984; Johnson *et al.* 1993; Norris & Georges 1993; Metcalfe-Smith 1994; Friedrich *et al.* 1996), usually organic pollution, and the inability to detect moderate degradation. Moreover, biotic indices should be used in conjunction with, and not as replacements for, conventional indices based on physical and chemical parameters.

A number of alternative approaches to the rapid bioassessment of river ecosystems from a community perspective have been pursued, but few have found as widespread application as biotic indices. This is mainly due to the relative difficulty of using the alternatives compared with biotic indices. The two approaches that have found increasing application are the 'multimetric (or composite) approach', represented by IBI (index of biotic integrity) and the 'multivariate approach' exemplified by RIVPACS (River Invertebrate Prediction And Classification System; Wright 1995; Wright *et al.* 2000).

SUMMARY AND CONCLUSIONS

1. Water quality indices (WQIs) are extensively used all over the world to abstract the numerical values of several water quality characteristics of a sample into a single value. Due to the ease with which WQIs enable an interpretation of the overall quality of water, they play a very important role in the monitoring, comparison and control of water quality. But the WQIs of the modern and post-modern times have been almost exclusively based on physical and chemical characteristics, and have seldom included 'biological' characteristics (other than BOD and faecal coliforms). This approach serves

a useful purpose, yet is inadequate in fully reflecting the status of a water body. This is due to the fact that WQIs based on physicochemical parameters are basically stressor-oriented; they operate on the principle of linking different stressors to possible biological responses. This predictive ability can, however, be precise only where a known cause–effect relationship exists between a specific stressor and the biological component. But such cause–effect relationships, for a specific suite of conditions, can at best be determined with laboratory bioassays under controlled conditions, and are not fully applicable in real-life situations.

2. This inherent limitation of stressor-oriented assessment makes it necessary to complement the conventional WQIs with response-based ones; the latter assess environmental health on the basis of the status of the responding organisms. They involve the monitoring of biological or ecological indicators present in a water body to gauge the response of the water body to a disturbance. The 'disturbance' can be any factor that disrupts the water body at ecosystem, community or population levels. It follows that the response-oriented approach indicates that something has or has not actually gone wrong in response to a stressor.

3. The stressor-oriented as well as the response-oriented approaches have obvious uses and specific benefits in water quality management. The current thrust is towards operationally integrating the two approaches so that the resulting methodology incorporates the benefits of both.

4. Biotic indices (BIs) have evolved from the response-based approach. In BIs, different taxa from a particular group of organisms are assigned different sensitivity weighting, or 'scores', based on the tolerance or sensitivity of different taxa to particular pollutants. The scores of all the individual taxa sampled at a site are summed and/or averaged to provide a value by which the ecological health of the biotic community, hence the health of the water body, is gauged. Some BIs include abundance estimates in the scoring system. A number of commonly used BIs have been described in the paper, highlighting their distinguishing features as well as limitations.

5. Benthic macroinvertebrates are the most widely used group in biotic indices but various other organisms have also been used including bacteria, protozoa, diatoms, algae, macrophytes, macroinvertebrates and fish.

The advantages with benthic macroinvertebrates are that they are largely non-mobile, ubiquitous and relatively abundant. There are often many species within a community with varying sensitivities to stresses and relatively quick reaction times. This provides a wide spectrum of graded and recognizable responses to environmental perturbation. The responses of many common macroinvertebrate species to different types of pollution have been extensively documented, which is an added advantage.

6. Even as biotic indices are very effective in complementing conventional WQIs, they are also subject to several limitations. The most serious one pertains to the difficulty in finding appropriate 'control' sites. Ideally a 'control' site should be truly reflective of natural, unpolluted conditions and thus serve as reference or 'control' with which the test sites can be compared to know whether a certain impact causes an aquatic assemblage or ecosystem to respond in some way that is outside the natural range of variation. Unfortunately, because of widespread human encroachments everywhere, it is very difficult to find non-impacted sites in most regions, especially in lowland areas, for use as 'control' sites. Consequently, minimally disturbed or the least-impacted of the available sites generally have to be used to determine the *best attainable* reference condition. Moreover, unlike WQIs based on physical and chemical parameters, BIs are unable to detect moderate degradation.

7. Several attempts to modify BIs and make their use simpler and faster have been made, but few have found as widespread application as BIs. The two approaches that have found increasing application are the 'multimetric (or composite) approach', represented by the Index of Biotic Integrity and the 'multivariate approach' exemplified by RIVPACS (River Invertebrate Prediction And Classification System). Neither is simpler or faster than any BI but both are arguably more comprehensive.

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