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Responses of freshwater biota to human disturbances: contribution of *J-NABS* to developments in ecological integrity assessments

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Abstract. Effective ecosystem management in the face of human alterations depends on our ability to quantify ecologically significant changes and to discriminate among impact levels and types. We reviewed the literature on biological responses of freshwater biota to human disturbances over the last century. Many of the main methods for assessing ecological integrity originated in Europe [e.g., Saprobien Index, Trent Biotic Index, Biological Monitoring Working Party, River InVertebrate Prediction And Classification System, multiple traits] or in North America [e.g., Index of Biotic Integrity, leaf-litter breakdown, functional feeding groups]. We used bibliometric data to detail the contribution of *J-NABS* to developments in assessment of ecological integrity. A total of 225 bioassessment articles were published in *J-NABS* from 1986 to 2007. This total was ~½ of the number of bioassessment articles published over the same period in *Freshwater Biology* (*FWB*; another leading freshwater journal used for comparison), but the proportion of bioassessment articles in the 2 journals was similar (~20%). The proportion of bioassessment articles in *J-NABS* has increased over the last 25 y. This trend indicates the growing interest in the topic and the delay (~30 y since the Clean Water Act) of scientific action in response to water legislation. Taxonomic composition and multivariate techniques, multimetric indices, lotic systems, macroinvertebrates, land use, and overall human impacts were major themes in *J-NABS* bioassessment articles. Progress in bioassessment is needed in 3 main areas. First, bioassessments done at large spatial scales are needed to meet requirements of new environmental policies. Second, bioassessment should shift from consideration of taxon losses to losses of ecological functions. Third, statistical techniques are needed for predictive assessment of deviation in ecological integrity between expected (natural) and observed (natural or impacted) conditions. Assessment of functional integrity based on multiple traits is a promising area because biological traits represent universal biological characteristics that are connected to ecosystem functions. However, the relationships between ecophysiological traits of individuals and ecosystem-scale responses must be properly defined for specific types of human impairment, and utility of the approach for routine bioassessments applied by managers must be tested.

Key words: ecological integrity, bioassessment, biological traits, ecosystem function, streams and lakes, review.

Use of biological signatures to assess anthropogenic ecological impairment is a well-established practice in freshwater ecology (Hynes 1960 [Fig. 1], Hellawell 1986, Cairns and Pratt 1993). Early in the 19th century, Europeans became aware of the poor condition of their rivers. For example, in Great Britain, the increase of human diseases associated with bad water quality led to specific legislation (e.g., Gas Works Clauses Act 1847, Salmon Fisheries Act 1861–1865, see Hynes 1960; see Fig. 1 for these and other key environmental policies). Therefore, early bioassessments used bacteriological methods to detect fecal and other organic pollution (Hynes 1960). Anthropogenic threats to water quality increased in developed countries and included salinization (~1900), metal pollution (~1910), eutrophication, NO_3^- pollution, contaminants (i.e., pesticides, insecticides associated with agricultural intensification, ~1930–1940), and acidification (associated with increased industrialization,

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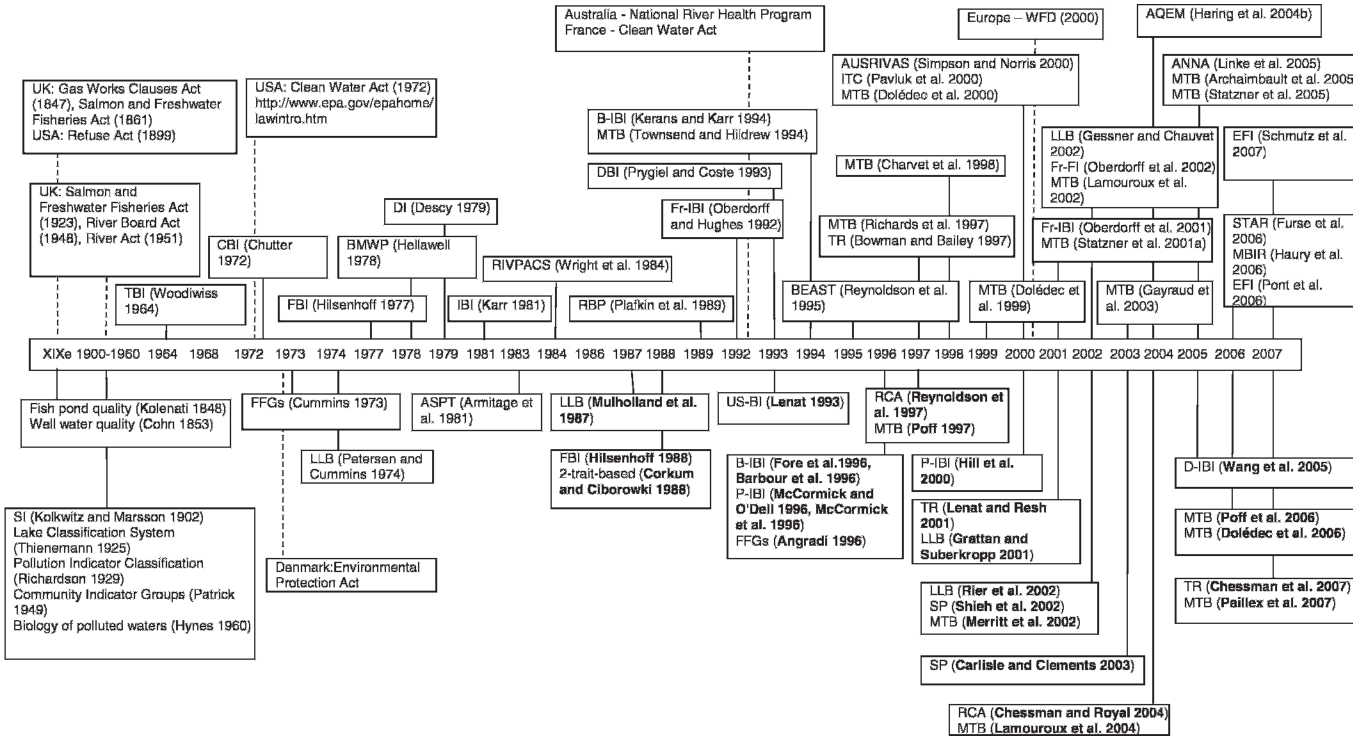

FIG. 1. Biological assessment time line. Dashed lines indicate environmental policies. Bold font indicates paper was published in *J-NABS*. LLB = leaf litter breakdown, SP = secondary production, FFG = functional feeding group, TR = taxonomic resolution, DK = Denmark. See Appendix for abbreviations for bioassessment approaches.

~1965) (e.g., Meybeck and Helmer 1989). Acceleration and accumulation of human pressures on water resources stimulated applied freshwater scientists to develop biomonitoring tools for detecting ecological impairment. Furthermore, increasing geographical extent of the impact of human activities led to studies of nonpoint source effects associated with landuse intensity at the catchment scale (e.g., Allan et al. 1997) and climate warming at a continental scale (e.g., Mouthon and Daufresne 2006). In addition, many human activities (e.g., aquaculture, cargo-ship traffic) facilitated dispersal of species beyond their natural ranges (reviewed in Kolar and Lodge 2001) and caused rising economic and ecological costs, which revived interest in the topic in freshwater research (e.g., Ricciardi and Rasmussen 1998, Clarke et al. 2004, Devin et al. 2005, Leprieur et al. 2008, Statzner et al. 2008).

The complexity of anthropogenic alterations of water resources (e.g., Karr and Chu 2000) promoted the concept of biological integrity, "the capability to support and maintain a balanced, integrated, adaptative community of organisms having species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Frey 1977, p. 128).

Aquatic flora and fauna integrate ecosystem changes over time and can reveal a source of pollution even if the source is not discharging pollutants at the time of sampling (e.g., Hynes 1960). Therefore, responses of organisms were used as surrogates for frequent water-chemistry measurements in assessment of biological integrity. Developments in bioassessment techniques led to the concept of ecological indicators, which are measurable characteristics of the structure or function of ecological systems (Niemi and McDonald 2004). Ecological indicators include landscape conditions, physical and chemical characteristics, biotic conditions (organisms, communities, and ecosystems), ecological processes (energy and material flow), and natural or anthropogenic disturbance regimes (USEPA 2002). The primary role of ecological indicators is to provide a measure of ecosystem responses to anthropogenic disturbances, i.e., of deviations from ecological integrity (Niemi and McDonald 2004). Ecological integrity has both structural and functional components (Minshall 1996). Structural integrity refers to the taxonomic composition of aquatic communities, whereas functional integrity refers to rates, patterns, and relative importance of biological processes. Obviously, structural and functional integrity are somehow linked (Cummins 1973; Fig. 1), but functional components have scarcely been considered in traditional assessments (e.g., Bunn and Davies 2000).

In our review, we consider the structural and functional component of biological assessment in a qualitative and quantitative analysis of the literature. The reference condition is the main topic of a separate article of this special issue (**Hawkins et al. 2010**[1]). Therefore, we focus on biological assessment of the functional integrity of freshwaters at the ecosystem and community level. In addition, we compare coverage of and impact on the topic of biological responses to human disturbances by the *Journal of the North American Benthological Society* (*J-NABS*) and *Freshwater Biology* (*FWB*), freshwater ecology journals with similar citation impact factors between 2002 and 2007 (*J-NABS*: 1.6–2.3, *FWB*: 1.6–2.8), as a means to indicate the relative contribution of *J-NABS* to developments in assessment of ecological integrity of freshwaters.

General Development of the Field

Assessment of structural integrity

Assessment of structural integrity is based on lists of taxa in biological assemblages. Taxon identity and combinations of taxa (metrics) or metrics (indices) are used to assess the biological condition of a water body relative to some reference condition (**Hawkins et al. 2010**).

Biotic indices.—Thieneman (1914) attributed the birth of biological monitoring to Kolkwitz and Marsson (1902; Fig. 1). These pioneers suggested the use of flora and fauna (i.e., the entire aquatic community) to assess freshwater health in the "Saprobien system" (Saprobien Index [SI]) (see Appendix for abbreviations for bioassessment approaches). This approach is a cornerstone in the development of biological indicators of human alterations of freshwater systems throughout the world, although research on the topic started earlier (e.g., Kolenati 1848, Cohn 1853; Fig. 1).

From this cornerstone arose an array of biotic indices based on formulae or standard tables (e.g., Trent Biotic Index [TBI], Woodiwiss 1964 [Fig. 1]; Tuffery and Verneaux 1968, Andersen et al. 1984, Lang et al. 1995) or on scoring systems, such as Biological Monitoring Working Party (BMWP; Hellawell 1978; Fig. 1) and Average Score Per Taxon (ASPT; Armitage et al. 1983; Fig. 1). Use of such Biotic Indices (BIs) for monitoring ecological conditions of freshwater systems spread all over the world (Fig. 2). Most were based on invertebrates (e.g., Sharma and Moog 1998), but diatoms also were used (e.g., Descy 1979, Prygiel and Coste 1993; Fig. 1). Haury et al. (2006; Fig. 1) designed the Macrophyte Biological Index for Rivers (MBIR), a biotic index based on the concept of trophic state (Thienemann 1925; Fig. 1) that uses macrophyte communities to assess river water quality. Today, most workers in European countries use BIs in their national surveys (e.g., Iversen et al. 2000), and workers in countries of other continents have designed their own BIs (Sharma and Moog 1998; Fig. 2). For example, Stark (1993) derived the New Zealand Macroinvertebrate Community Index (MCI) from the BMWP scoring system, and Chutter (1972; Fig. 1) created the South African Chutter Biotic Index (CBI) from SI (Fig. 2).

The Water Framework Directive (WFD; Fig. 1), which is largely based on the reference condition approach (see **Hawkins et al. 2010**), arose from a growing need for convergence in environmental policies across the European Union (European Commission 2000). WFD provides a framework to harmonize traditional assessment protocols of individual countries in Europe through intercalibration so that various national assessment systems have comparable integrity scales (e.g., Furse et al. 2006 [Fig. 1], Hawkins 2006). For example, Sandin and Hering (2004) intercalibrated national indices at the European level and showed that ASPT accurately identified good quality sites for all river types (with one exception), whereas SI misclassified many sites. They concluded that the definition of quality classes at the European scale must consider natural differences between river types and must correct for national differences in the perception of ecological quality. The harmonization approach compares national biological assessment procedures with external benchmarking data sets, such as Assessment system for the ecological Quality of streams and rivers throughout Europe using benthic Macroinvertebrate (AQEM) data (Hering et al. 2004b; Fig. 1). National quality-class boundaries are then redefined until differences between national and benchmarking samples become statistically insignificant (Buffagni et al. 2006, Buffagni and Furse 2006).

North American freshwater ecologists were reluctant to use the SI system because it was based on European species, assessed only one stressor (organic pollution) among multiple potential stressors, and relied on the indicator species concept, which overlooks the natural causes of population variations (Cairns and Pratt 1993). This a priori rejection of the indicator species concept probably delayed the development of BIs in North America (Cairns and Pratt 1993), where earlier suggestions for use of BIs in

[1] Boldface indicates paper was published in *J-NABS*

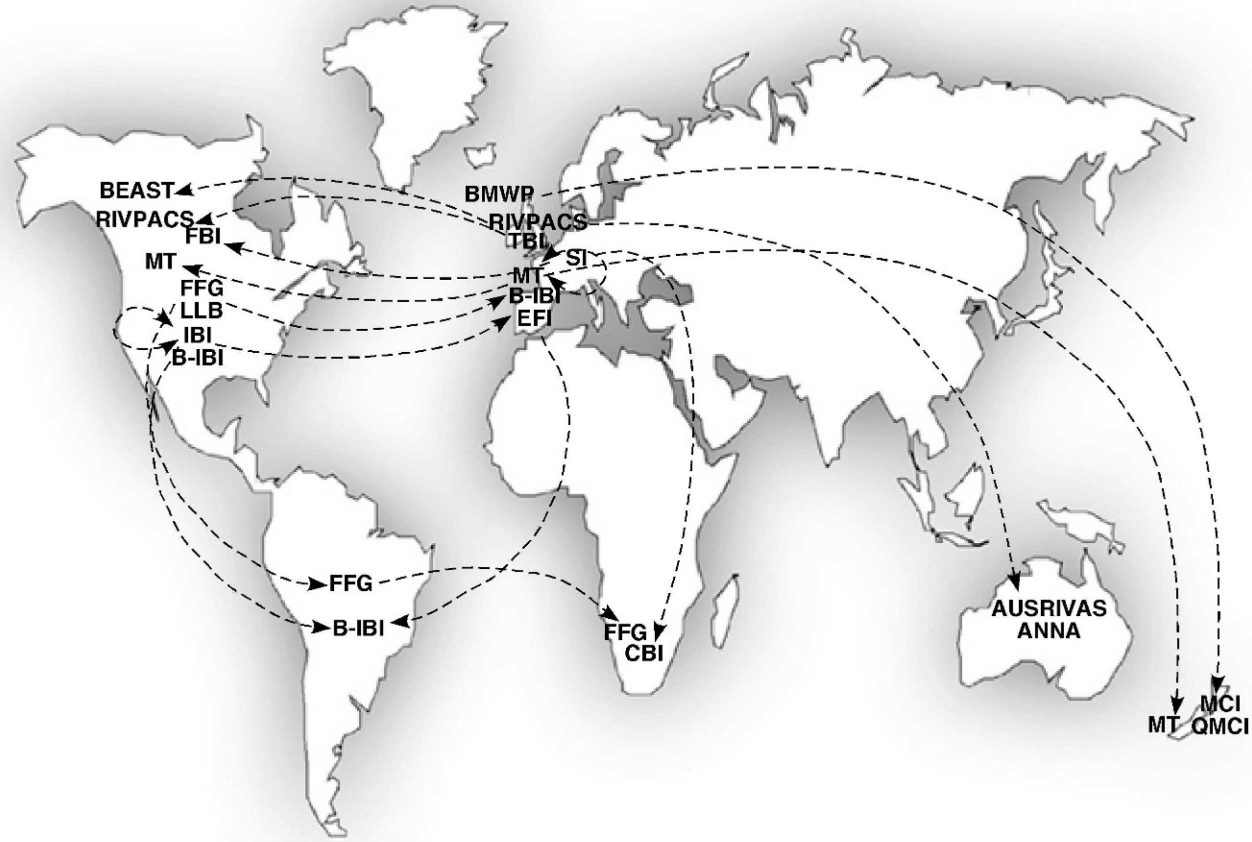

FIG. 2. Movements of selected bioassessment techniques from point of origin across the world. LLB = leaf litter breakdown, SP = secondary production, FFG = functional feeding group. See Appendix for abbreviations for bioassessment approaches.

the US (Richardson 1929; Fig. 1) had been ignored (but see Hilsenhoff 1977; Fig. 1).

J-NABS published the Family Biotic Index (FBI, **Hilsenhoff 1988**; Fig. 1) and the BI for southeastern US streams (US-BI, **Lenat 1993**; Fig. 1), which strongly depended on organisms' tolerances and indicator values. BIs have been addressed in a significant proportion of *J-NABS* papers (~12%; Table 1), but workers in the US preferred to incorporate BIs as a component of multimetric approaches (Hawkins 2006).

Multivariate approach.—The increasing calculation power of computers and related statistical developments for synthesizing taxonomic composition or taxonomic completeness led to predictive biological assessment tools in the 1980s. These developments were rooted in multivariate techniques, such as Principal Components Analysis (PCA; Pearson 1901) and Correspondence Analysis (CA; Hirschfeld 1935), that had been designed decades earlier and influenced freshwater research in the 1970s (e.g., Lévêque and Gaborit 1972, Ibanez and Seguin 1972). Subsequent developments in multivariate analyses included Canonical Correspondence Analysis (CCA; ter Braak 1986), which was widely used in ecological (Birks et al. 1996) and freshwater (ter Braak and Verdonschot 1995, Houghton 2007) sciences, and coinertia analysis (COIA), which was developed to relate freshwater species composition to environmental conditions (Dolédec and Chessel 1994) and has a wide range of applications (Dray et al. 2003).

Multivariate analyses are used widely for determining patterns of assemblage structure, inferring species–environment relationships, and assessing human impacts on ecosystems (**Cao et al. 2002**). Despite initial skepticism related to the complex outputs of multivariate analyses (**Norris 1995**), ~30% of *J-NABS* bioassessment articles were based on the approach (a proportion comparable to that in *FWB*; Table 1). Some investigators consider these techniques exploratory and useful only in case of limited knowledge of a given ecological system (e.g., **Fore et al. 1996**; Fig. 1). Others regard them as useful tools in bioassessment because they can be used to determine the most important variables that influence community composition (e.g., Houghton 2007) and to describe and predict the effects of human disturbance on aquatic biota (e.g., Mykra et al. 2008). CCA or COIA currently are used to identify sites with different levels and types of pollution (e.g., **Zamora-Munoz and Alba-Tercedor 1996**, Truu et al. 2002, Hicham and Lofti 2007), assess the effects of land use on aquatic biota (e.g., Aznar et al. 2004, **Kratzer et al. 2006**, Couceiro et al. 2007), design new metrics or indices for diagnosing environmental stressors (Griffith et al. 2001, Haase and Nolte 2008), and validate metric performance (Prygiel et al. 1999).

A prominent example of the use of multivariate techniques in biological assessment is the River InVertebrate Prediction and Classification System (RIVPACS, Wright et al. 1984; Fig. 1), which uses reference conditions (**Reynoldson et al. 1997** [Fig. 1], Stoddard et al. 2006, but see **Chessman and Royal 2004** [Fig. 1]) to compare a predicted minimally disturbed macroinvertebrate assemblage with an observed assemblage (**Hawkins et al. 2010**). This comparison results in an observed/expected (*O/E*) ratio of a given biological variable (e.g., BMWP, ASPT), which lies on a scale from impacted ($O/E <$ 0.50; Wright et al. 2000) to undisturbed ($O/E = 1.00$) conditions.

RIVPACS was developed for streams and rivers in the UK, but the RIVPACS method has been promoted in the US (Hawkins et al. 2000), implemented in other countries (e.g., AUStralian RIVer Assessment Scheme [AUSRIVAS], Turak et al. 1999, Simpson and Norris 2000 [Fig. 1]; Assessment by Nearest Neighbor Analysis [ANNA], Linke et al. 2005 [Fig. 1]), or extended to lakes (BEnthic Assessment of SedimenT [BEAST], Reynoldson et al. 1995; Fig. 1) (Fig. 2). These methods rely on reference conditions, so their application to other regions requires redefinition of the reference condition for the new geographic area (e.g., Alba-Tercedor and Pujante 2000) because biases can occur in the predictions when test sites are not compared with reference sites of the same region (e.g., **Yuan et al. 2008**). Separating biological responses to natural and anthropogenic causes at greater spatial scales can be difficult with RIVPACS-like methods because of missing environmental variables in the redefinition of the reference condition, insufficient number of reference sites, and insufficient replication of tested ecological conditions (**Hose et al. 2004**, Hawkins 2006). Current RIVPACS-like models cannot identify specific stressors (e.g., acidification, logging; Bonada et al. 2006), although combining data from multiple types of organisms (e.g., benthic macroinvertebrates, fish, macrophytes, diatoms) could help solve this problem. This strategy corresponds to the notions of early freshwater ecologists that the entire aquatic biocenosis should be used when assessing ecosystem conditions (Kolkwitz and Marsson 1902, Patrick 1949 [Fig. 1], Hynes 1960). For example, in European rivers, benthic diatom assemblages have a greater sensitivity to eutrophication or organic pollution than do other assemblages (Hering et al. 2006b, Johnson et al. 2006b, Feio et al. 2007), a feature also observed in the Everglades where periphyton was the most sensitive group of organisms to P loading (**McCormick et al. 1996**). Furthermore, responses of various assemblages to specific stressors are influenced by stream type; e.g., benthic invertebrates were better suited for assessing hydromorphological degradation in smaller streams (Feio et al. 2007), whereas fish or macrophytes were equally suited (with invertebrates) in larger lowland streams (Hering et al. 2006b, Johnson et al. 2006b).

ter Braak and Juggins (1993) developed a weighted averaging partial least squares (WA-PLS) approach for predicting environmental conditions from biota (calibration). This technique has been used in paleolimnological research to reconstruct past environmental conditions from biotic remains (e.g., Gasse et al. 1995). This technique provides an indication of how present reference conditions match historical ones, and can be especially useful when depicting past nutrient loading. Paleolimnology is used primarily in lakes and is little mentioned in *J-NABS* (<1% of bioassessment articles; Table 1).

Multimetric approach.—In his history of water law in North America, Karr (1991) indicated that early legislation (Refuse Act of 1899; Fig. 1) focused on growing problems of disease and oil pollution in navigable waters, whereas awareness of restoration and maintaining the biological integrity of the US waters started with the 1972 Water Pollution Control Amendment (WPCA, Clean Water Act; Fig. 1). Karr (1981; Fig. 1) initiated the first studies based on the multimetric index (MMI) approach in which attributes (metrics) of fish assemblages were used to evaluate human effects on a stream and its watershed. Metrics are biological measures representing aspects of the composition, function of assemblages, or other characteristics of individuals that should vary with increased human impact. The first fish Index of Biotic Integrity (IBI) considered 12 metrics, such as species richness and composition, trophic composition, abundance, and health condition. Metrics are rated relative to what would be expected at an undisturbed site in a stream of similar size in the same region. The sum of these ratings provides an IBI value that is regarded as an integrative and quantitative measure of local biological integrity (Karr 1991) or overall human impact. The original fish IBI was developed for streams in the US Midwest, but it has been adapted for use in other regions by modification or replacement of metrics (e.g., Miller et al. 1988, Steedman 1988).

The multimetric approach is used with lotic invertebrates throughout the US (e.g., Invertebrate Community Index [ICI], Ohio EPA 1987; Benthic Index of Biotic Integrity [B-IBI], Kerans and Karr 1994 [Fig. 1], Barbour et al. 1995), and has been adapted to periphyton (**McCormick and O'Dell 1996**, **McCormick et al. 1996**, **Hill et al. 2000**; Fig. 1) and diatom assemblages (**Wang et al. 2005**; Fig. 1). The Diatom IBI (D-IBI) is based primarily on community composition metrics (e.g., richness and composition, absolute or relative abundances of dominant taxa, tolerance/intolerance of taxa) and de-emphasizes individual-level measures (e.g., disease, tumors, fin damage, skeletal anomalies). The Periphyton Index of Biotic Integrity (P-IBI) includes periphyton-specific metrics of ecological function, such as % motile diatoms and phosphatase activity (**Hill et al. 2000**).

Many papers based on MMI developments have been published in *J-NABS* (~13% of bioassessment articles; Table 1). Several of these papers, such as Barbour et al. (**1996**; Fig. 1), Fore et al. (**1996**), and McCormick et al. (**1996**) have been among the 100 most-cited *J-NABS* articles. MMI development provided a common biological approach to assessing the condition of most rivers in the US. By the middle 1990s, multimetric biological assessment was used in 42 US states and was under development in 6 states (Karr et al. 2000). However, regional differences in the fauna or flora and in the intensity of stress at human-impacted sites generally make comparison of MMIs difficult at large spatial scales (Hawkins 2006).

The multimetric approach spread from North America to Europe (Fig. 2) where it currently is under evaluation as a tool for assessing river conditions with invertebrates (e.g., Hering et al. 2004a, Johnson et al. 2006a, Feld and Hering 2007) and fish (Oberdorff and Hughes 1992, Oberdorff et al. 2002, Pont et al. 2006, Schmutz et al. 2007; Fig. 1). However, the approach is not easily transferable because metrics are selected based on their apparent sensitivity to region-specific stressors (e.g., Norris and Hawkins 2000). In contrast to indices used in the US, the European index covers the continent (Pont et al. 2006, Schmutz et al. 2007) rather than regions and includes functional metrics (e.g., reproduction and trophic guild for fish, Pont et al. 2006; feeding habits and ecological preferences for macroinvertebrates, Hering et al. 2006a). Hering et al. (2006a) used a method for designing MMIs that excluded numerically unstable metrics, used correlations between appropriate metrics and a selected anthropogenic impact gradient, and included only the most robust metrics (those with the best response to the anthropogenic impact gradient), given the need to balance the 4 types of metrics. The need for such a selection process was recently outlined by Stoddard et al. (**2008**) in a large-scale assessment using MMI. The scale of variation of metrics is defined according to reference conditions, and the MMI is generated by combining the selected metrics (e.g., 12 metrics for Böhmer et al. 2004), which represent a small proportion of the metrics tested (165 metrics were computed by Böhmer et al. 2004). This method reflects the strong empirical basis of many European metrics, and is in contrast with the use of theoretical or conceptual understanding of biologically important attributes to select metrics (e.g., Norris and Hawkins 2000, Hawkins 2006).

According to their proponents, multimetric approaches provide biologically interpretable handles (metrics) that better facilitate decision-making than do methods based on indices and that generate reliable indices of the overall impact of human activities (Barbour et al. 1999). However, the meaning of IBIs has been questioned (e.g., no predictability, no diagnostic power, no real reason for high or low index values; Suter 1993), as has the validity of summing heterogeneous metrics into a single measure of river condition (effects on one metric could be blurred by effects on other metrics; Suter 1993, Norris and Hawkins 2000).

Partialling out the effects of natural variation on assemblages should improve the ability of metrics to indicate human impacts on freshwater communities (Pan et al. 2004, Feld and Hering 2007). For example, Moya et al. (2007) elaborated models describing metric responses to natural gradients in the absence of any significant disturbance. They used the residuals of each metric model to select metrics that most effectively discriminated between reference and disturbed sites. In a last step, they converted residuals of selected metrics into probabilities and added the probabilities to obtain an MMI.

The discriminatory capacity of metrics and rates of misclassification of disturbed sites as reference sites (type II error) can vary according to the group of aquatic organisms used. For example, landuse inten-

Table 1. Approaches, systems, biological models, and stressors addressed by bioassessment articles published in *J-NABS* and *Freshwater Biology* (*FWB*). Articles associated with several categories were counted more than once. Numbers in parenthesis are percentages of the total number (from 1986–2007) of bioassessment articles in each journal in decreasing order of representation in *J-NABS*.

	1986–1989		1990–1993		1994–1997		1998–2001		2002–2005		2006–2007		1986–2007	
Categories	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*
Approaches														
Ecological function														
Abundance, diversity (biomass)[a]	7	20	2	16	12	18	15	40	17	30	9	23	62 (27.6)	147 (27.1)
Leaf litter breakdown	3	1	–	6	–	4	1	2	5	11	1	7	10 (4.4)	31 (5.7)
Functional feeding groups	–	2	1	7	3	6	1	5	5	8	–	8	10 (4.4)	36 (6.6)
Ecosystem metabolism[b]	–	5	–	2	–	5	–	4	6	5	3	9	9 (4.0)	30 (5.5)
Food web (stable isotopes)	–	–	–	–	1	–	3	1	4	3	–	8	8 (3.6)	12 (2.2)
Multiple biological traits	–	–	–	–	–	1	–	3	1	4	3	1	4 (1.8)	9 (1.7)
Taxonomic composition														
Abundance, diversity (individuals)	5	25	6	28	15	30	29	52	36	70	21	55	112 (49.8)	260 (48.0)
Multivariate[c]	–	7	1	18	11	24	12	35	23	44	22	41	69 (30.7)	169 (31.2)
Multimetric indices	–	–	1	–	5	2	9	7	7	7	7	8	29 (12.9)	24 (4.4)
Biotic indices	1	–	2	2	4	1	4	4	8	9	7	8	26 (11.6)	24 (4.4)
Paleolimnology	–	3	–	4	1	8	–	10	–	10	1	4	2 (0.9)	39 (7.2)
Population														
Genetic diversity	1	–	1	–	2	1	1	2	1	3	–	6	6 (2.7)	12 (2.2)
Biotic interactions	–	3	–	2	2	3	–	11	–	8	–	9	2 (0.9)	36 (6.6)
Organism/suborganism														
Individual traits[d]	4	10	4	19	4	12	14	15	10	29	10	31	46 (20.4)	116 (21.4)
Cell processes[e]	–	–	–	3	6	9	4	6	5	8	3	3	18 (8.0)	29 (5.4)
Bioassays	–	3	–	4	6	1	–	2	–	4	–	1	6 (2.7)	15 (2.8)
Fluctuating asymmetry[f]	–	–	1	–	–	3	1	2	1	1	–	–	3 (1.3)	6 (1.1)
Systems														
Lotic	11	30	7	43	33	53	38	60	57	83	31	74	177 (78.3)	343 (63.3)
Lentic	1	27	3	24	12	22	11	48	10	41	11	30	48 (21.3)	192 (35.4)
Others[g]	–	1	–	1	–	1	–	4	3	3	–	8	3 (1.3)	18 (3.3)
Biological models														
Benthic macroinvertebrates[h]	9	21	8	32	34	31	39	41	49	56	35	58	174 (77.3)	239 (44.1)
Periphyton	2	6	1	5	14	10	15	11	16	19	10	13	58 (25.8)	72 (13.3)
Fish	–	8	–	8	6	6	4	17	8	24	3	25	21 (9.3)	88 (16.2)
Bacteria	1	5	–	2	–	1	3	3	5	1	1	9	10 (4.4)	22 (4.1)
Phytoplankton	–	13	–	12	1	2	1	10	1	17	2	9	5 (2.2)	82 (15.1)
Riparian vegetation	2	1	1	1	1	1	–	2	–	4	1	4	5 (2.2)	18 (3.3)
Zooplankton	–	10	–	9	1	3	1	13	–	16	2	7	4 (1.8)	58 (10.7)
Macrophytes[i]	–	9	–	5	–	1	–	5	3	8	1	12	4 (1.8)	55 (10.1)
Fungi	1	–	–	–	–	–	–	–	3	5	–	5	4 (1.8)	11 (2.0)
Amphibians	–	–	–	–	1	2	–	–	1	1	1	2	3 (1.3)	5 (0.9)
Bryophytes	–	1	–	–	–	–	2	5	1	–	–	3	3 (1.3)	9 (1.7)
Birds	–	1	–	1	–	–	–	2	–	1	–	3	–	8 (1.5)

TABLE 1. Continued.

Categories	1986–1989		1990–1993		1994–1997		1998–2001		2002–2005		2006–2007		1986–2007	
	J-NABS	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*	*J-NABS*	*FWB*
Stressors addressed														
Land use[j]	–	2	–	10	2	10	4	19	17	20	11	24	34 (15.1)	85 (15.7)
Overall impacts	–	2	–	3	5	7	8	10	10	18	9	11	32 (14.2)	51 (9.4)
Exotic species	1	–	–	1	4	4	7	11	9	16	4	10	25 (11.1)	42 (7.7)
Contaminants[k]	–	7	1	10	6	11	3	5	7	12	1	4	18 (8.0)	49 (9.0)
Physical habitat[l]	–	5	–	12	4	7	5	16	5	21	3	39	17 (7.6)	100 (18.5)
Water chemistry	1	11	3	6	–	1	3	4	3	15	7	12	17 (7.6)	49 (9.0)
Eutrophication	1	18	–	15	2	18	6	36	5	15	3	24	17 (7.6)	126 (23.2)
Logging	2	1	1	3	3	1	3	4	6	5	1	4	16 (7.1)	18 (3.3)
Reservoir effects[m]	4	4	1	5	3	1	1	5	5	2	1	4	15 (6.7)	21 (3.9)
Climate change[n]	–	–	1	1	2	5	3	12	4	13	2	12	12 (5.3)	43 (7.9)
Sewage[o]	1	1	–	–	2	4	1	7	3	2	3	6	10 (4.4)	20 (3.7)
Acidification	2	14	1	16	2	12	3	9	–	12	1	4	9 (4.0)	67 (12.4)
Others[p]	–	3	1	1	–	–	2	3	1	2	–	–	4 (1.8)	9 (1.7)
Salinization	–	1	–	1	–	–	–	2	–	4	1	6	1 (0.4)	14 (2.6)

[a] Includes secondary production

[b] Includes respiration, microbial activity

[c] Includes RIVPACS, BEAST, AUSRIVAS, ANNA and other multivariate techniques (principal components analysis, correspondence analysis, canonical correspondence analysis) (see Appendix for bioassessment method abbreviations)

[d] Includes size, growth, fecundity, settlement, diet, filtration rate, mobility, mortality, uptake per chlorophyll *a*

[e] Includes biomarkers, algal pigments, nitrification potential, bioaccumulation, stoichiometry (C:N, C:P)

[f] Morphological deformities

[g] Includes groundwaters, wetlands

[h] Includes meiofauna

[i] Includes seed banks

[j] Includes urbanization

[k] Includes heavy metals, arsenic toxicity, polycyclic aromatic hydrocarbons, fuel spills, mining activities, pesticides, insecticides

[l] Includes morphology (e.g., bridges, channelization, lake shore), flow regulation (includes storm water release, water abstraction, interbasin water diversion)

[m] Includes impoundment release

[n] Includes global warming, ultraviolet radiation effects

[o] Includes organic pollution, waste water

[p] Includes fish release, mussel harvesting, recreational activities (including boat traffic)

sity and nutrient enrichment were well detected (classification error < 15%) by metrics derived from diatom and benthic invertebrate assemblages. In contrast, metrics derived from fish and aquatic macrophyte assemblages performed better (lower type II error) as indicators of physical alterations than of nutrient enrichment (Johnson et al. 2006a).

Assessment of functional integrity

Functional integrity can be assessed directly by measuring ecosystem metabolism (Newbold et al. 1981, **Bott et al. 2006**), leaf litter breakdown (LLB; Kaushik and Hynes 1971, Gessner and Chauvet 2002 [Fig. 1]), and secondary production (SP; Benke 1993, Buffagni and Comin 2000), or indirectly via functional feeding groups (FFGs; Cummins 1973) and 2- or multiple-trait-based (MTB) approaches (**Corkum and Ciborowski 1988**, Statzner et al. 2001a; Fig. 1). Functional integrity has also been assessed via single traits (e.g., growth, mortality, dispersal rates) in bioassay or toxicity tests (e.g., Buikema and Voshell 1993).

Direct assessment of ecosystem processes (LLB and SP).—Ecosystem-level processes that are potentially useful in stream assessment include SP (e.g., **Benke and Huryn 2010**), sediment and community respiration, rate of nutrient removal, and LLB (Gessner and Chauvet 2002).

LLB is a central process in the exchange of matter between terrestrial and aquatic ecosystems (e.g., **Sponseller and Benfield 2001, Lamberti et al. 2010**). LLB coefficients (*k* values) represent rates of decomposition in an exponential decay model (Petersen and Cummins 1974; Fig. 1). Gessner and Chauvet (2002) suggested that *k* values from standardized leaf packs could be used in bioassessments as: 1) ratios of *k* values between impacted and reference sites, 2) absolute *k* values if no reference is available, and 3) ratios of *k* values between coarse- and fine-mesh bags to test the relative contribution of shredders and microorganisms to LLB.

LLB is affected by modifications of riparian and catchment vegetation, nutrient enrichment, and urbanization (e.g., Webster and Benfield 1986, **Meyer et al. 2005**). LLB rates decrease in response to mine pollution (Niyogi et al. 2001, Lecerf and Chauvet 2008) and acidic atmospheric deposition (Dangles et al. 2004, and references therein). When N and P concentrations are low, nutrient enrichment can increase LLB rates via increased fungal biomass and sporulation rates (**Grattan and Suberkropp 2001**; Fig. 1). However, high organic and inorganic nutrient concentrations can depress conidia production by aquatic hyphomycetes and increase macroinvertebrate density, which can accelerate LLB rates (Pascoal et al. 2003). More research on fungal decomposers is needed to better understand underlying LLB mechanisms (**Pascoal et al. 2005**).

Agricultural land use can have both positive (nutrients) and negative (sedimentation) effects on LLB (Niyogi et al. 2003). Hagen et al. (**2006**) found similar LLB rates across landuse categories from forested to heavily cultivated areas. Pascoal et al. (**2005**) failed to detect differences in LLB rates between a nutrient-enriched site and its upstream reference site, but impairment was obvious when rates were compared between fine- and coarse-mesh bags. These results suggest that metrics based on ratios of *k* values between fine- and coarse-mesh bags might be more suitable than metrics based on ratios of *k* values between reference and impacted sites.

One of the first papers showing a strong decrease in LLB in response to acidification was published in *J-NABS* (**Mulholland et al. 1987**; Fig. 1), but papers describing use of LLB rates as biological response to human disturbance are infrequent in *J-NABS* (4.4%, Table 1). For example, Grattan and Suberkropp (**2001**) is the most-cited (44 citations in the Web of Science® database at the Institute for Scientific Information [ISI] Web of Knowledge℠ site [ISI-Web®]; http://apps.isiknowledge.com/, checked in June 2008) of these articles but is not among the 100 most-cited *J-NABS* articles. Changing water chemistry (including nutrient loading) and land uses have been the main stressors addressed by LLB assessments, but recent research has addressed more subtle ecosystem modifications. For example, LLB has been used as an early warning indicator of alteration of riparian vegetation by invasive plant species, which indirectly affect instream communities (e.g., Lecerf et al. 2007). An innovative study published in *J-NABS* used LLB to assess the effect of global change on ecosystem processes (**Rier et al. 2002**; Fig. 1). This study showed that an increase of atmospheric CO_2 levels might increase litter recalcitrance to microbial activity and decrease LLB rates, which would decrease availability of C and N, thereby affecting food webs (**Kominoski et al. 2007**).

SP, the formation of heterotrophic biomass over time, is another important measure linking populations (biomass, growth rate, survivorship, development time) and communities to ecosystem-level processes (e.g., energy flow, Benke 1993, **Benke and Huryn 2010**). Buffagni and Comin (2000) suggested that calculating SP for dominant taxa in the different habitats should be an effective means to assess the ecological integrity of mountain streams. However, annual SP can vary naturally by a factor of ~4 over short distances in lake outflows (Statzner 1987) or in forested streams in response to the relative composition of litter patches (**Kobayashi and Kagaya 2004**). Such variability would limit its application at larger scales (see Statzner and Lévêque 2007 for large-scale SP variability).

SP responds to ecosystem degradations, such as river regulation (Raddum and Fjellheim 1993), biological invasions (Statzner and Lévêque 2007), changing riparian forests (Whiles and Wallace 1997), logging (Kedzierski and Smock 2001), stream acidity (Thomsen and Friberg 2002), stream contamination (**Carlisle and Clements 2003**; Fig. 1), organic pollution (Statzner and Lévêque 2007), and multiple sources (**Shieh et al. 2002** [Fig. 1], **de Lange et al. 2004**). Whole-stream annual macroinvertebrate production was greater in logged than in unlogged stream sections (Kedzierski and Smock 2001) and was lower before than after clearcutting based on changing energy resources (Stone and Wallace 1998). Such changes generally are associated with changes in FFG composition. Invertebrate SP was lower in streams draining pine monocultures than in undisturbed hardwood catchments (Whiles and Wallace 1997). Metal contamination reduced invertebrate abundances and SP (**Carlisle and Clements 2003**). In contrast, biological invasions or organic pollution caused an increase of SP (Statzner and Lévêque 2007). In cases of multiple sources of impairment, Shieh et al. (**2002**) attributed increased invertebrate SP to high nutrient availability and low predation pressure in urban and agricultural sites, whereas de Lange et al. (**2004**) suggested that moderate levels of sediment contamination scarcely affects benthic SP.

Carlisle and Clements (2005) measured several ecosystem functional responses (LLB, microbial respiration, and shredder SP) to metal contamination. The species most sensitive to metal contamination was a functionally dominant species. Thus, loss of a single species might impair ecosystem function. In contrast, de Lange et al. (**2004**) observed that contamination of sediment by trace metals and other chemicals affected invertebrate community structure, but SP of oligochaetes and chironomids was not correlated with the level of contamination.

Papers published in *J-NABS* have contributed significantly to our understanding of fundamental aspects of SP (e.g., **Benke et al. 1999** with 200 ISI-Web® citations). However, only one paper published in *J-NABS* has applied SP to bioassessment (**Carlisle and Clements 2003**). This paper has received 21 citations and is not among the 100 most-cited *J-NABS* articles.

Indirect assessment of ecosystem processes based on invertebrate FFGs.—A significant part of the annual LLB in streams is caused by invertebrate activities, e.g., >25% of LLB in a headwater stream could be attributed to shredder activities (Cuffney et al. 1990). Organic matter cycling and energy flow are basic processes associated with the functional integrity of lotic ecosystems (Vannote et al. 1980). Thus, benthic macroinvertebrates often are assigned to FFGs based on feeding mechanisms, energy sources, and food size (Cummins 1973).

Bioassessments with FFGs can be based on absolute or relative abundance (of individuals, biomass, or SP) of a single FFG. For example, Kedzierski and Smock (2001) reported higher SP of filtering and gathering collectors in a logged than in an unlogged stream section, whereas SP of other FFGs changed little with logging. In contrast, grazers dominated macroinvertebrate communities in catchments with recently logged forests, whereas the proportion of shredders increased with time since logging (Nislow and Lowe 2006). Urbanization and associated changes in stream quality also can affect FFGs. For example, proportions of collectors and gatherers increased, whereas proportions of filter-feeders, scrapers, and shredders decreased with watershed imperviousness associated with urbanization (Stepenuck et al. 2002). In another study, higher proportions of scrapers occurred at urban sites, whereas higher proportions of filter-feeders occurred at agricultural sites (Bacey and Spurlock 2007). Agricultural activities increase sediment deposition and silting, which can decrease absolute densities of all FFGs while favoring relative densities of gatherers (Rabeni et al. 2005) or filter-feeders (**Dolédec et al. 2006**; Fig. 1). Other stressor types are poorly detected by FFGs. For example, a strong salinity gradient caused only slight alterations of trophic structure (Piscart et al. 2005).

Ratios of abundances of ≥ 2 FFGs have been used as surrogates of ecosystem processes. For example, Merritt et al. (**2002**; Fig. 1) suggested using the ratio of total shredders to total collectors as a surrogate for the ratio of coarse particulate organic matter (CPOM) to fine particulate organic matter (FPOM) and the ratio of filtering collectors to gathering collectors as a surrogate for the ratio of suspended particulate organic matter (SPOM) to benthic particulate organic matter (BPOM). Paillex et al. (**2007**; Fig. 1) used such ratios as surrogates for top-down control of assemblage structure and habitat stability. Ratios declined along a gradient from disconnected to more connected sites in a river floodplain, a result that suggested the ratios might be useful for monitoring conditions in floodplain water bodies after connectivity between main and abandoned channels was restored (**Paillex et al. 2007**). Compin and Cereghino (2007) reported higher proportions of gathering collectors but lower proportions of shredders and filtering collectors in human-impacted landscapes than in natural ones.

Pavluk et al. (2000; Fig. 1) designed an index of trophic completeness (ITC), based on 12 trophic groups defined by combinations of diet composition, feeding mechanisms, food size, food-acquisition behavior, and energy transfer through prey ingestion. According to Pavluk et al. (2000), all 12 FFGs should occur in undisturbed rivers, and the absence of $\geq$1 FFGs indicates an effect of anthropogenic activities. Bij de Vaate and Pavluk (2004) confirmed that disturbances cause loss of specific trophic groups. They suggested the ITC might have large-scale applicability because of its independence from distance to source, geographical characteristics, and seasonality, but acknowledged that the ITC was limited in its ability to identify specific stressors. The approach requires more testing because the current reference condition is defined from samples of stony substrate in a few unpolluted river sites.

FFGs have been useful for bioassessments in the northern hemisphere, but generally have failed to detect anthropogenic effects in the southern hemisphere (Palmer et al. 1996, Thorne and Williams 1997). In Argentina, most FFGs (except shredders) occurred along the entire river despite modifications of the hydrological regime caused by an artificial reservoir (Miserendino 2007). FFG-based metrics did not distinguish reference from disturbed sites in Bolivian rivers (Moya et al. 2007) because of a high proportion of generalist feeders that easily switched among food resources (**Wantzen and Wagner 2006**). Another problem is that the FFG to which many southern hemisphere taxa belong is not known; thus, authors often assign taxa to FFGs based on North American FFG classifications. Analyses of gut contents and mouthpart morphologies of southern hemisphere taxa would solve this problem (e.g., Tomanova et al. 2006).

The evidence that FFGs are surrogates for ecosystem processes has been questioned (Mihuc 1997). A generalist feeding habit apparently is a common strategy among invertebrates in lotic systems. Moreover, FFGs group invertebrates on the basis of the size, rather than the type, of the particles they process as food (Cummins and Klug 1979), and FFGs do not indicate whether processed particles are assimilated (Mihuc 1997).

The FFG approach originated in the US, but the most cited *J-NABS* bioassessment article based on FFG as a biological response to human disturbance (**Angradi 1996**; Fig. 1) received 36 citations and does not appear among the 100 most-cited *J-NABS* articles. Thus, based on citation reports, papers published in *J-NABS* have contributed relatively little to the development of the topic.

Indirect assessment of ecosystem processes based on multiple biological traits of invertebrates.—A recent approach to bioassessment combines FFGs with a wide variety of other biological traits including food types, body size, fecundity, voltinism, and dispersion ability of benthic macroinvertebrates to describe functional community structure (e.g., **Corkum and Ciborowski 1988** [2 traits], Charvet et al. 1998 [Fig. 1], Minshall and Robinson 1998, Snook and Milner 2002, **Lamouroux et al. 2004** [Fig. 1], **Dolédec et al. 2006**).

In its simplest definition, a trait is an approximation for organism performance, and the change of this performance might scale up to ecosystem functioning as follows: responses of ecophysiological and life-history traits of individuals (e.g., growth, fecundity, survival) might affect demographic traits of populations (e.g., birth, death, emigration, immigration), which in turn, might affect community structure and dynamics (e.g., species occurrence) and ecosystem functioning (e.g., energy flow, chemical cycling) (Violle et al. 2007).

Multiple-trait based (MTB) assessments are deeply rooted in ecological theory (Statzner et al. 2001b) and rely on predicting ecological responses to selection induced by environmental conditions (i.e., habitat templet concept, Southwood 1977, 1988, Townsend and Hildrew 1994 [Fig. 1], **Poff 1997** [Fig. 1]). Use of multiple biological traits has several advantages over traditional assessments based on taxonomic composition. Traits are expressed in many species, so functional community descriptions can be compared among regions that differ in their taxonomic composition and might have large-scale applicability (Statzner et al. 2001a, Horrigan and Baird 2008). Biological trait responses of invertebrate assemblages to similar reference environmental conditions converge at the scale of regions (Archaimbault et al. 2005; Fig. 1), nations (Charvet et al. 2000), continents (Statzner et al. 2004, 2005 [Fig. 1]), and the world (Statzner et al. 1997, see Lamouroux et al. 2002 [Fig. 1] for fish trait response at this scale).

Most traits should be affected predictably by various types of human impact (e.g., Statzner et al. 2005, **Dolédec et al. 2006**, Dolédec and Statzner 2008). For example, flow increases should select for smaller size, firmer attachment to substrate, and streamlined shape, whereas flow decreases should favor organisms with larger size, good swimming abilities, and spherical shape. Siltation should favor traits that enable penetration of fine substrates (e.g., burrowing), whereas traits that render organisms susceptible to smothering by sediment (e.g., lack of egg protection) should decrease. Organic pollution and resulting O_2 deficits should increase the proportion of organisms with aerial respiration. Heavy metal pollution should decrease organisms with smaller body size (greater body surface–volume ratio) or gill respiration (increase of body surface) and predators (bioaccumulation of toxins). MTB approaches might be able to identify the most sensitive traits for a given impact type and facilitate targeted management actions.

Preliminary studies have shown the responses of traits to various human disturbances. For example, Richards et al. (1997; Fig. 1) proposed models in which catchment- and reach-scale physical characteristics were used to predict the occurrence of specific life-history and behavioral traits as a tool for landuse management. Charvet et al. (1998) reported that effluents from a wastewater treatment plant significantly changed many biological traits of benthic invertebrates in a small stream (e.g., more larger animals with fewer descendants and greater longevity downstream of the effluent). Statzner et al. (2001a) found differences in the trait compositions of communities in near-pristine and sewage-impacted habitats at the European scale. Invertebrate traits discriminated overall human impact on river reaches in the large European rivers (Dolédec et al. 1999 [Fig. 1], Usseglio-Polatera and Beisel 2002) and significantly discriminated among various levels of human impacts (Gayraud et al. 2003; Fig. 1). The frequently debated topic of best taxonomic level for bioassessment (e.g., Bowman and Bailey 1997, **Lenat and Resh 2001**, **Chessman et al. 2007**; Fig. 1) also was assessed for MTB approaches. Trait descriptions of genera provided an accurate picture of the functional community structure at different spatial scales in reference situations (Dolédec et al. 2000; Fig. 1), and trait weighting by the presence–absence of genera reliably discriminated various levels of human impacts (Gayraud et al. 2003).

Multivariate techniques have been designed to analyze relationships between biological traits and environmental conditions weighted by taxon presence–absence or abundance (Dolédec et al. 1996, Legendre et al. 1997). Mellado Diaz et al. (2008) recently used the 3-table ordination (RLQ) technique to illustrate that more invertebrates with small size, multivoltinism, diapause, or ovoviviparity were present in frequently disturbed streams in semiarid areas than in more stable streams in undisturbed forested upland areas.

The quality of MTB assessments depends largely on data quality and availability as for other biomonitoring tools (e.g., Hering et al. 2004a, Stoddard et al. 2008). A major problem of MTB approaches is to assemble consistent trait information because knowledge is often lacking, especially for rare taxa. Macroinvertebrate trait databases have been developed in Europe (e.g., Usseglio-Polatera et al. 2000a, b, Tachet et al. 2002, Gayraud et al. 2003, Ilg and Castella 2006, Statzner et al. 2007), the US (Vieira et al. 2006, Bêche and Resh 2007), and in the southern hemisphere (**Dolédec et al. 2006**, Tomanova and Usseglio-Polatera 2007). Most of these databases use a fuzzy coding procedure (Chevenet et al. 1994) to address trait variability that often is related to differences among life stages of a taxon, differences across the range of a taxon, or absence of solid knowledge for a taxon. Trait databases for biota other than macroinvertebrates have been developed for zooplankton (Barnett et al. 2007), aquatic macrophytes (Willby et al. 2001), and fish species in Europe (Pont et al. 1995, Santoul et al. 2005, Blanck et al. 2007) and in the US (Goldstein and Meador 2004, 2005). MTB assessments potentially are applicable to ecosystems other than freshwaters, including transitional waters (Mouillot et al. 2006) and marine environments (e.g., Bremner et al. 2003, 2006, Frid et al. 2008).

MTB approaches are widely used in Europe, but have received limited attention in other parts of the world (e.g., **Merritt et al. 2002**, Horrigan and Baird 2008). Less than 2% of *J-NABS* bioassessment papers address MTB approaches (Table 1), but an often-cited (>200) *J-NABS* article by Poff (**1997**) has promoted use of biological traits as responses to natural environmental filters. The most-cited *J-NABS* article dealing with MTB bioassessment (**Merritt et al. 2002**) has received 15 citations and is not among the 100 most-cited J-NABS articles.

Contribution of *J-NABS* to the Evolution of Bioassessment

Coverage of topic over time

From the first (1986) to the latest (2007) issues that we analyzed, *J-NABS* published 225 articles (22% of the total articles published in *J-NABS* during the period) and *FWB* published 541 articles (21% of the total articles published in *FWB*) on the response of freshwater biota to human disturbance. Authors of 17.3% of *J-NABS* bioassessment articles were from outside North America, whereas in *FWB*, this proportion was ~4× higher (66.4%). The proportion of total *J-NABS* articles that addressed bioassessment has increased significantly from ~10% in the mid-1980s to the present-day level of >30% ($R^2 = 0.701$, $p < 10^{-5}$; Fig. 3). The proportion of bioassessment articles

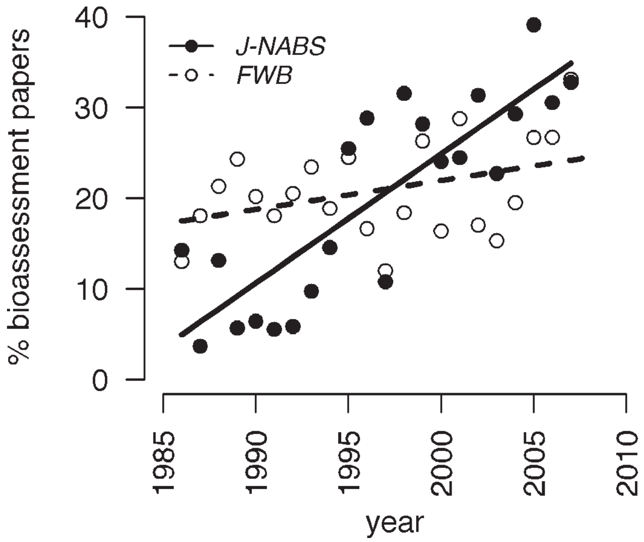

Fig. 3. Percentage of total articles published in *J-NABS* ($R^2 = 0.701$, $p < 10^{-5}$) and in *Freshwater Biology* (*FWB*) ($R^2 = 0.155$, $p = 0.07$) that addressed bioassessment.

in *FWB* tended to increase from the mid-1980s until the mid-1990s and then increased significantly from 1995 to 2007 ($R^2 = 0.39$, $p < 0.03$; Fig. 3). This increase might be attributable to the release of the Applied Issues section of the journal in 1994 (Hildrew and Townsend 1994).

Four main approaches appear in *J-NABS* and *FWB* bioassessment articles in similar proportions. These approaches include abundance and diversity of individuals, multivariate techniques, abundance and diversity in terms of biomass and production, and use of a single trait as an assessment target (Table 1). The proportion of bioassessment articles dealing with MMIs and BIs was much higher in *J-NABS* than in *FWB* (Table 1), whereas the reverse occurred for studies on biotic interactions and paleolimnology. The proportion of bioassessment studies based on FFGs was lower in *J-NABS* than in *FWB* (Table 1), although pioneers in the field were North Americans.

Far more bioassessment studies in *J-NABS* addressed lotic than lentic systems (Table 1). Other systems (ground waters or wetlands) were rarely addressed in *J-NABS*, although submissions of manuscripts on these systems is encouraged in the Instructions for Authors of the journal. Bioassessment articles in *J-NABS* mainly considered benthic macroinvertebrates and periphyton, and occasionally fish, as biological models (Table 1). The two latter models were rare in studies before 1994 and gained in importance relative to studies based on invertebrates only after 1994 (Table 1). Bioassessment based on riparian vegetation and indicators of ecosystem functioning (e.g., bacteria, fungi) were rarely addressed in *J-NABS* (Table 1). Differences in coverage of systems (e.g., lotic vs lentic) or biological models (e.g., periphyton vs phytoplankton) between *J-NABS* and *FWB* are related primarily to the different foci of the journals (benthos vs fresh waters).

The stressors most frequently addressed in *J-NABS* bioassessment articles (>10% of bioassessment articles) were land use, overall human impacts, and exotic species (Table 1). Stressors, such as eutrophication, physical habitat, acidification, climate change, and salinization, were addressed less frequently in *J-NABS* than in *FWB*, whereas the reverse occurred for overall impacts, logging, and reservoir effects. The frequency of bioassessment studies addressing land use (including urbanization) began increasing in 1986 and 2000 in *FWB* and *J-NABS*, respectively (Table 1).

J-NABS *citation impacts and key bioassessment articles*

We checked the 100 most-cited articles (on all topics) published in *J-NABS* and *FWB* in the ISI-Web® (checked in June 2008) to assess the citation impact of bioassessment articles. Nineteen of the 100 most-cited *J-NABS* articles addressed some aspect of bioassessment. A methods article on the reference condition approach (**Reynoldson et al. 1997**) has the highest number of citations (162; Table 2). *J-NABS* bioassessment articles with >100 ISI-Web® citations addressed use of macroinvertebrate assemblages to design a BI and assign water-quality ratings (**Lenat 1993**) or a regional index of stream condition (B-IBI; **Barbour et al. 1996**), and to compare the relative merits of multimetric and multivariate approaches (**Fore et al. 1996**). Articles with ≥74 citations (Table 2) addressed effects of eutrophication and landuse activities on periphyton communities (**McCormick et al. 1996**, **McCormick and O'Dell 1996**, **Pan et al. 1996**), and the response of macroinvertebrate assemblages to metal pollution (**Clements 1994**). Articles with ≥59 citations (Table 2) addressed effects of exotic species on aquatic communities (**Silver Botts et al. 1996**, **Charlebois and Lamberti 1996**, **Stewart et al. 1998**, **Strayer 1999**). Other frequently cited articles (46–62 citations; Table 2) addressed the effects of nutrient enrichment on algal communities (**Biggs 2000**, **Dodds and Welch 2000**, **Hill et al. 2000**), effects of downstream physical disturbance on upstream catchments (**Pringle 1997**), and aspects of sampling and identification protocols (**Kerans et al. 1992**, **Hannaford and Resh 1995**, **Lenat and Resh 2001**).

Twenty-three of the 100 most-cited *FWB* articles addressed some aspect of bioassessment. An article on the effects of catchment land use on stream

TABLE 2. The most frequently cited bioassessment articles (among the 100 most-cited articles of each journal), with their citations in each journal (i.e., *J-NABS* articles cited in *J-NABS* papers, *Freshwater Biology* [*FWB*] articles cited in *FWB* papers) and in the Science Citation Index (ISI-Web®) available by June 2008, and their numerical ranking in each list. *J-NABS* had 19 bioassessment articles among the 100 most cited; FWB had 23 bioassessment articles among the 100 most cited.

	Year	Journal-specific citations	Rank	ISI-Web® citations	Rank
J-NABS					
Reynoldson et al.	1997	16	1	162	1
Lenat	1993	10	6	136	2
Fore et al.	1996	9	8	128	3
Barbour et al.	1996	16	1	118	4
McCormick et al.	1996	5	15	96	5
McCormick and O'Dell	1996	3	17	89	6
Clements	1994	7	11	87	7
Pan et al.	1996	12	4	74	8
Strayer	1999	4	16	73	9
Silver Botts et al.	1996	6	13	70	10
Biggs	2000	3	17	62	11
Hill et al.	2000	8	9	61	12
Pringle	1997	2	19	61	12
Charlebois and Lamberti	1996	8	9	59	14
Lenat and Resh	2001	12	4	59	14
Stewart et al.	1998	7	11	59	14
Dodds and Welch	2000	6	13	56	17
Kerans et al.	1992	10	6	48	18
Hannaford and Resh	1995	14	3	46	19
FWB					
Allan et al.	1997	20	7	176	1
Hall and Smol	1992	16	12	167	2
Biggs and Close	1989	27	2	146	3
Biggs	1995	19	9	126	4
Moss et al.	1987	25	5	119	5
Richards et al.	1997	30	1	116	6
Jeppesen et al.	2000	19	9	111	7
Ormerod et al.	1987	27	2	106	8
Reeders et al.	1989	7	19	103	9
Parsons and Norris	1996	24	6	97	10
Horner et al.	1990	11	15	92	11
Ormerod and Edwards	1987	26	4	91	12
Sponseller et al.	2001	6	22	84	13
Richards et al.	1993	10	17	81	14
Townsend et al.	1997	14	13	79	15
Bunn et al.	1999	12	14	78	16
Winterbourn	1990	20	7	72	17
Irvine et al.	1989	17	11	72	17
Behmer and Hawkins	1986	11	15	67	19
Oberdorff et al.	2001	8	18	66	20
Jones and Jugins	1995	7	19	66	20
Hart	1988	7	19	64	22
Fritz et al.	1993	6	22	63	23

integrity (Allan et al. 1997) has the highest number of citations (176; Table 2). The next most-cited article (167 citations; Table 2) is a paleolimnological study that inferred past total P from diatom remains (Hall and Smol 1992). Articles with $\geq$92 citations (Table 2) addressed periphyton responses to flow and nutrient disturbance (Biggs and Close 1989) and catchment land use (Biggs 1995), a topic to which the less frequently cited article by Horner et al. (1990) is closely linked. Other highly cited articles (97–119 citations; Table 2) addressed RIVPACS-like methods (Moss et al. 1987, Parsons and Norris 1996), use of a MTB approach to assess the effects of catchment land use on macroinvertebrate assemblages (Richards et al. 1997), lake eutrophication (Jeppesen et al. 2000), effects of acidification on macroinvertebrate assemblages (Ormerod et al. 1987), and effects of exotic species on lake water quality (Reeders et al. 1989). Articles with ≥78 citations (Table 2) addressed responses of stream macroinvertebrate communities at the catchment scale (Ormerod and Edwards 1987, Richards et al. 1993, Sponseller et al. 2001, Townsend et al. 1997) or responses of ecosystem measures (i.e., gross primary production and respiration) to catchment disturbance (Bunn et al. 1999). Two articles with 72 citations (Table 2) addressed eutrophication in ponds (Irvine et al. 1989) and streams (Winterbourn 1990). Articles with ≤67 citations (Table 2) addressed topics, such as the response of macroinvertebrate production to opening of river canopy (Behmer and Hawkins 1986), modeling fish species distribution to elaborate the basis of a national MMI (Oberdorff et al. 2001; Fig. 1), inferring lake trophic status from sediment-core diatom assemblages (Fritz et al. 1993, Jones and Juggins 1995), or the effects of turbidity on lake zooplankton feeding rates (Hart 1988).

In summary, number and biomass (or production) of individuals, followed by multivariate analyses, MMIs, BIs, and FFGs, were the main approaches used in the most frequently cited *J-NABS* bioassessment articles (Table 3). The most frequently cited *FWB* articles used number of individuals less often. In both *J-NABS* and *FWB*, lotic systems, benthic macroinvertebrates, and periphyton were leading categories (Table 3). Articles on stressors, such as overall human impacts, water chemistry, and exotic species were more frequent among the most-cited bioassessment articles in *J-NABS*, whereas articles on acidification and eutrophication were more frequent among the most-cited bioassessment articles in *FWB* (Table 3).

The ISI-Web® citation ranking of the 19 most-cited bioassessment articles was not related to the *J-NABS* citation ranking (Table 2), suggesting that *J-NABS* bioassessment articles receiving top citations by authors publishing in *J-NABS* did not have the same value for authors of papers in the general scientific literature. The 4 most-cited *J-NABS* bioassessment articles that were most frequently cited in the ISI-Web® are conceptual or concern very general methods applicable in places other than North America. In contrast, the 4 *J-NABS* bioassessment articles most frequently cited in *J-NABS* were articles on specific concerns about sampling accuracy (**Hannaford and Resh 1995**) and taxonomic identification levels (**Lenat and Resh 2001**), which interest primarily benthic macroinvertebrate ecologists. Moreover, all but one of the first authors of the 19 most frequently cited *J-NABS* bioassessment articles were North Americans, whereas only ⅓ of the first authors of the most

Table 3. Distribution of the most frequently cited bioassessment articles published by *J-NABS* (19) and *Freshwater Biology* (*FWB*) (23) (see Table 2) across geographic origin of the first author (North America [NA] vs nonNA), and types of approaches, systems, biological models, and stressors. Numbers in parentheses are percentages of total bioassessment articles published in the journal within a category.

Category	*J-NABS*	*FWB*
Geographic origin		
NA	18 (94.7)	8 (34.8)
NonNA	1 (5.3)	15 (65.2)
Approaches		
Ecological function		
Abundance, diversity (biomass)	7 (36.8)	9 (39.1)
Functional feeding groups	3 (15.8)	1 (4.3)
Ecosystem metabolism	–	1 (4.3)
Food webs (stable isotopes)	–	1 (4.3)
Multiple biological traits	–	1 (4.3)
Taxonomic composition		
Abundance, diversity (individuals)	12 (63.2)	7 (30.4)
Multivariate	8 (42.1)	7 (30.4)
Multimetric indices	5 (26.3)	2 (8.7)
Biotic indices	3 (15.8)	–
Paleolimnology	–	2 (8.7)
Population		
Genetic diversity	1 (5.3)	–
Biotic interactions	–	1 (4.3)
Organism/suborganism		
Individual traits	1 (5.3)	2 (8.7)
Cell processes	1 (5.3)	1 (4.3)
Systems		
Lotic	15 (78.9)	16 (69.6)
Lentic	5 (26.3)	7 (30.4)
Biological models		
Benthic macroinvertebrates	13 (68.4)	12 (52.2)
Periphyton	8 (42.1)	3 (13.0)
Fish	3 (15.8)	2 (8.7)
Phytoplankton	1 (5.3)	3 (13.0)
Zooplankton	–	3 (13.0)
Macrophytes	–	1 (4.3)
Riparian vegetation	–	1 (4.3)
Stressors addressed		
Overall impacts	6 (31.6)	4 (17.4)
Eutrophication	4 (21.1)	8 (34.8)
Exotic species	4 (21.1)	–
Water chemistry	3 (15.8)	1 (4.3)
Logging	1 (5.3)	1 (4.3)
Land use	1 (5.3)	1 (4.3)
Contaminants	1 (5.3)	–
Acidification	–	2 (8.7)
Physical habitat	–	1 (4.3)

frequently cited 23 *FWB* bioassessment articles were North Americans (Table 3).

We compared the 19 most-cited *J-NABS* with 19 most-cited *FWB* bioassessment articles (after randomly omitting 4 from the 23 *FWB* articles). We found fewer ISI-Web® citations to *J-NABS* than to *FWB* articles (Kruskal–Wallis test, $p < 0.04$; *J-NABS* median = 70 citations, *FWB* median = 91 citations). We assumed that this difference was related to journal age because we expected an increase of the total citations with the time elapsed since publication. However, we did not find such a trend for the most-cited bioassessment articles in *J-NABS* ($p > 0.25$) or *FWB* ($p > 0.85$). However, ISI-Web® citations did increase significantly with the age of the article when all 225 *J-NABS* bioassessment articles were considered ($R^2 = 0.22$, $y = -2.44x + 4919$, $p < 10^{-12}$) and when all 541 *FWB* bioassessment articles were considered ($R^2 = 0.19$, $y = -1.52x + 3054$, $p < 10^{-15}$). **Resh and Kobzina** (**2003**) found only 1 bioassessment article (Plafkin et al. 1989; Fig. 1) among the 10 most-cited articles in *J-NABS* over the period 1995 to 2000. The percentage of *J-NABS* bioassessment articles cited >12 times (a threshold used by **Resh and Kobzina 2003**) over the period 1986 to 2007 was 42.6%, whereas the percentage of *FWB* bioassessment articles cited >12 times was 52.4%.

Thus, ISI-Web® citations indicate that *J-NABS* has not contributed as strongly as *FWB* to the bioassessment topic, and to date, *J-NABS* has contributed weakly to the debate about functional integrity.

Future Directions of the Topic

Biological assessment of freshwater systems has been done for >100 y and the science has matured. The science is rooted in Europe, but bioassessment techniques for assessing both structural and functional integrity have been dispersed across the world in many forms (Fig. 2). From early empirical indices, freshwater scientists developed new or improved existing biomonitoring tools in response to the increasing importance of ecosystem health in all parts of the world and associated new demands of policymakers to enhance ecological functions across large geographical areas. We analyzed hundreds of articles for this review, and we think that 3 major points should be addressed in the future. First, bioassessment of fresh waters should be able to discriminate between natural and human-induced changes at both local and global scales (Ormerod et al. 1999). Many nations have developed nation-specific bioassessment techniques, but comparison of data across nations is difficult because of differences in field methods or indicators used (e.g., Furse et al. 2006, Hawkins 2006), and variability in sample identification (Stribling et al. 2008) that might be caused by compromises made between scientific rigor and practical implementation (e.g., **Hughes and Peck 2008**). Second, bioassessment of fresh waters should rely on prediction (e.g., Norris and Hawkins 2000) and should address deviations in the ecological integrity as the difference between expected (natural) and observed (natural or impacted) conditions (**Reynoldson et al. 1997**, Wright et al. 2000, Nijboer et al. 2004, Hawkins 2006). Third, bioassessment of fresh waters should use both occurrence of taxa and derived metrics (structural integrity) and measures of functions of ecological systems (Dale and Beyeler 2001).

MTB approaches are taxon-free metrics that quantify evolutionary responses to selective environmental forces across large geographic areas. Both physiological and life-history traits might control species composition and relative abundances (Robson et al. 2005) and affect ecosystem properties (Violle et al. 2007). MTB approaches are indirect assessments of ecosystem processes, so they should be tested in parallel with direct functional measurements to clarify their reliability for depicting functional stream impairment (e.g., Lecerf et al. 2006). Further studies also are needed to determine the reliability of a given combination of biological traits for assessing specific human impacts in a context of multiple stressors.

According to Poff et al. (**2006**; Fig. 1), MTB approaches should focus on evolutionarily labile traits that have low statistical correlations (i.e., are phylogenetically independent) and are linked in a mechanistic way to a specific human impact to avoid confounding effects of trait syndromes (see also Horrigan and Baird 2008). This argument calls for constructing molecular (e.g., **Ball et al. 2005**) or at least morphometric phylogenies of freshwater invertebrates and other groups to isolate the part of history that explains the presence of a given taxon in a given environment. Another approach, recently proposed by Verberk et al. (2008a) outlined 13 life-history strategies based on biological traits of macroinvertebrates, their interrelation known from theory, and their functional implications. Verberk et al. (2008b) applied their approach to lentic waters and argued that the observed difference in strategies across water bodies are easily interpretable and mechanistically explained. Reducing multiple biological traits to a lower number of life-histories limits the potential of the approach to indicate specific human impacts in the context of multiple stressors.

Many trait databases have been developed independently in various countries with little communication among researchers. This pattern of development resembles that which occurred for biotic indices and could lead to negative consequences, such as one trait database per group or per researcher or the formation of schools or camps, situations that have delayed development of bioassessment tools in the past (Bonada et al. 2006). The compatibility and strength of the various trait databases that have been developed in various countries should be investigated. We think construction of a collective database accessible to all researchers is appealing and could ensure rigor, for example, in the way traits are coded (Statzner et al. 2007).

Use of biological traits, rather than species richness metrics, enables consideration of functional biodiversity (e.g., Solow and Polasky 1994, Petchey and Gaston 2002). The disadvantage of richness as a measure of biological diversity is that all taxa are taken into account on an equal basis regardless of their abundance, their biological characteristics, or their function in the ecosystem. The problem of abundance can be circumvented by use of diversity and equitability indices. However, functional redundancy (caused by evolutionary constraints) and biological divergence or convergence across taxa (associated with life history or physiology) are rarely taken into account when assessing biological diversity of freshwater ecosystems (e.g., Bady et al. 2005, Heino 2008). Accounting for such functional redundancy could be done in 2 ways. First, trait clustering could be used to define functional groups (i.e., macroinvertebrate genera with similar biological or ecological traits) at a broad spatial scale (e.g., Europe; Usseglio-Polatera et al. 2000a). To our knowledge, the utility of number or diversity of functional groups as responses to human alteration has rarely been investigated (but see Heino 2005). Devin et al. (2005) considered the diversity (Shannon) of functional groups as a measure of functional diversity and recorded changes in the values of a functional diversity index in response to changing abundances of several exotic species. Second, measures of biological diversity that incorporate dissimilarities might be useful. Work in marine environments has considered the response of taxonomic distinctness to increasing environmental stress (Warwick and Clarke 1995), and some methods for measuring biological dissimilarities have been rediscovered recently (Ricotta 2005). Bady et al. (2005) weighted biological trait dissimilarities of stream invertebrate taxa by their abundances and obtained a biological diversity index they regarded as a surrogate for functional diversity. The index appeared to be independent of sampling effort but must be further tested against various types of human alteration across large geographic areas. Reducing the information from multiple biological traits into one trait diversity value per community can provide only a general indication of overall functional impairment. Identification of the stressor(s) causing the impairment will require more detailed analyses and should be facilitated by the multitude of biological traits (i.e., the multitude of biological sensors) so far described for many groups of freshwater organisms.

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APPENDIX. Acronyms used for bioassessment methods.

Abbreviation	Term	Abbreviation	Term
ANNA	Assessment by Nearest Neighbor Analysis	ITC	Index of Trophic Completeness
AQEM	Assessment system for the ecological Quality of streams and rivers throughout Europe using benthic Macroinvertebrates	MBIR	Macrophyte Biological Index for Rivers
ASPT	Average Score Per Taxon	MCI	Macroinvertebrate Community Index
AUSRIVAS	AUStralian RIVer Assessment Scheme	MMI	Multimetric Index
BEAST	BEnthic Assessment SedimenT	MTB	Multiple Trait-Based
BI	Biotic Index	P-IBI	Periphyton Index of Biotic Integrity
B-IBI	Benthic Index of Biotic Integrity	QMCI	Quantitative Macroinvertebrate Community Index
BMWP	Biological Monitoring Working Party	RBP	Rapid Bioassessment Protocol
CBI	Chutter Biotic Index	RCA	Reference Condition Approach
DI	Diatom Index	RIVPACS	River InVertebrate Prediction And Classification System
D-IBI	Diatom Index of Biotic Integrity	SI	Saprobien Index
EFI	European Fish Index	STAR	STAndardisation of River classifications
FBI	Family Biotic Index	TBI	Trent Biotic Index
FFG	Functional Feeding Group	TR	Taxonomic Resolution
Fr-FI	French Fish Index	US-BI	Southern United States Biotic Index
Fr-IBI	French Index of Biotic Integrity	WA-PLS	Weighted Averaging Partial Least Squares
IBI	Index of Biotic Integrity	WFD	Water Framework Directive
ICI	Invertebrate Community Index	WPCA	Water Pollution Control Act