

# Chapter 19

## Biomonitoring and Bioassessment



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### 19.1 Introduction

The water that we use, the air that we breathe, and the energy that we consume are limited resources. Among these, “water issues are one of the major problems that humanity must solve for its survival.” This maxim was a key conclusion reached by top-level decision-makers at the first Asia Pacific Water Summit in December 2007, marking the first time in history that all Asian states met to discuss water issues. This statement does more than characterize the Asian situation. It applies to our entire globe. Water managers and scientists are aware that sustainably managing a fundamental resource like water requires rigorous scientific data and analysis to understand aquatic ecosystem functioning. Proper sustainable management requires that we know the “quantity” and “quality” of a water source. This chapter describes the basics of the “qualitative aspects” of water monitoring and management, namely, the biomonitoring and the assessment of “river quality.”

Following centuries of human impacts on aquatic resources, monitoring of water chemistry has become a customary practice in many countries. The fundamental scientific aspects of sustainable water management in many areas of the world are still quite poorly understood, and biological monitoring tools are lacking for many developing and transitional countries (Bere and Nyamupingidza 2013). In recent decades, attempts to develop monitoring methodologies are increasing rapidly and have generated biological assessment and monitoring of aquatic resources that are reliable enough to be included in state monitoring programs in Europe, the United States, Australia, and South Africa. Following on this successful trend is a wave of

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increasing activity in developing, adapting, and testing of aquatic biomonitoring methods in Africa, Asia, and South America.

The need for aquatic biomonitoring is obvious, as rivers in most parts of the world are extremely overexploited and impacted in manifold ways. Assessing the status of rivers and identifying the threats is essential to develop adequate restoration and protection strategies. Each river or river reach is characterized by a unique signature of different biological and ecological characteristics as well as a large variety of pressures and impacts. Such a signature cannot be documented sufficiently by physical-chemical monitoring only. Biological monitoring covers a larger spectrum of pressures and multiple spatial scales over a longer time span. State-of-the-art environmental monitoring combines chemical and biological indicators in assessing the ecological conditions of aquatic systems.

## 19.2 History of Water Quality Assessment

### **The Ecological Assessment of River Quality Is Older than the Science of Ecology**

There is a simple answer to the question “Since when do we need water quality assessment?”: since humans destroyed their surface waters in a way that deteriorated drinking water quality. The first written record of water pollution was given about 350 years before Christ, when Aristotle reported on “black mud” and “red tubes”—as he called it—growing out of a “white slime” in brooks of the city Megara polluted by domestic sewage (Thienemann 1912). The famous Greek philosopher was the first who linked human pressures with observations of oxygen reduction (black decaying mud), a community of *Beggiatoa* sulfur bacteria (white slime), oligochaete sludge worms, and chironomids (red tubes). Aristotle’s knowledge fell into oblivion, and the beginning of water quality assessment had to wait for about 1800 years. Anyhow, observations of a correlation between the composition/distribution of certain aquatic invertebrate species and different water pollution levels are not very recent findings. One could even say that this knowledge is older than “ecology” itself (as defined by Ernst Haeckel in 1866). As early as Kolenati (1848), it was already concluded by F. A. Kolenati that the absence of caddis larvae from a stream can be caused by the presence of factories upstream (*Stettiner Entomologische Zeitung* 9). Triggered by the severe cholera epidemics in Europe, two researchers, A. H. Hassal, London (1850), and F. Cohn, Breslau (1853), discovered and published the relationship between organic pollution, river fauna, and the quality of drinking water based on bioindicators. In the United States, the earliest biomonitoring research originates from Forbes (1887) who invented the biological community concept. Basically, using this concept plant and animal communities of a river were used to assess the degree of organic pollution. Around 1900 two German scientists (R. Kolkwitz & M. Marsson) studied polluted rivers around Berlin and described defined communities of organisms in different zones of organic enrichment. They developed the concept of “biological indicators of pollution” in their so-called saprobic system, which is still in use in several Central and Eastern European states.

## 19.3 The Saprobic System

### Reliable and True and Frequently Used

Based on earlier research before the turn of the century, Kolkwitz and Marsson (1902) introduced the terms “Saprobien” for waste water organisms and “Katharobien” for organisms in clean rivers. From the stressor’s point of view, saprobity is the state of the water quality resulting from organic enrichment as reflected by the species composition of the community. Kolkwitz and Marsson published indicator lists for benthic algae and invertebrates, which served as a valuable tool for water quality assessment for some decades.

The saprobic system was adapted after World War II by Liebmann (1951) who published a widespread manual on saprobiology and provided a substantial list of indicators. On this basis he introduced water quality mapping, which permits the visualization of a river’s ecological status as “color-banded.” The power of such mapping techniques to convey complex information in a convincing way is evident in how it has stimulated politicians, decision-makers, water managers, and other stakeholders and the interested public to combat pollution. The acceptance of the saprobic system was increased remarkably by the development of the saprobic index by Pantle and Buck (1955) enabling quantification of pollution intensity. This index ranging from 1 (very good quality) to 4 (extremely poor quality) could be easily interpreted by the end users. At about the same time in the United States, Beck (1954) created a biotic index to provide a simple measurement of stream pollution and its effects on the biology of the stream. This development very likely happened independently, because both authors did not cite each other’s papers. Zelinka and Marvan (1961) modified the saprobic index by including the concept of saprobic valencies. They introduced a system reflecting the 100% (often bell-shaped) occurrence of a taxon among the water quality classes, i.e., ten points substituting the 100% are distributed among the four water quality classes. In Sládeček (1973) Sládeček summarized the knowledge in his book “System of Water Quality from the Biological Point of View” which served the following decades as a methodological bible for saprobiologists. In this time, the saprobic system was widely used in Central and Eastern Europe (e.g., Austria, Bulgaria, Czech Republic, Germany, Hungary, Romania, Slovakia, Slovenia, and former Yugoslavia).

The last update of the saprobic approach was precipitated around the millennium, when the European Water Framework Directive 2000/60/EC (WFD) substantially changed the biomonitoring approach of European aquatic ecosystems. Since then, the ecological status of water bodies needs to be defined based on type-specific approaches and reference conditions. Several countries decided to integrate the saprobic approach into the new integrative methodology for defining the ecological status of water bodies and thus to adjust the saprobic system. The revisions comprised alterations and additions to the list of indicator taxa, type-specific saprobic reference conditions (Rolauuffs et al. 2004) and an adaptation to the ecological status classification of the WFD. Currently, the saprobic system is part of the multi-metric indices used in Austria (Ofenböck et al. 2004, 2010a), Czech Republic (Kokes et al. 2006), and Germany (Meier et al. 2006).

## 19.4 Biotic Indices and Scoring

### Hundreds of Ways Toward One Goal

Since the early stages of water quality evaluation, some hundreds of methods for biological river status assessment have been developed (Birk et al. 2012). Unfortunately, the mathematical terms “index” and “score” in river status assessment are often used in a confusing way. A *biotic index* is a numerical expression of the sensitivity or tolerance of organism assemblages to anthropogenic stress. A *score* is a numeric expression of the ecological indicator status that can be used to calculate an index, which can be generated, e.g., as an average of scores of several indicators. The principle of biotic indices is to assign different types of taxa to different levels of disturbance. Sensitive taxa decrease or disappear, and tolerant taxa emerge or increase under stress.

The first indices were nearly simultaneously developed in the United States and Europe around 1950 (Beck 1954; Pantle and Buck 1955). The Trent Biotic Index (developed by Woodiwiss 1964, 1978) is seen as the origin of many biotic indices that are not following the saprobic approach, e.g., the “Indice biotique” (IB) in France (Verneaux and Tuffery 1967), the Belgian Biotic Index (BBI) (De Pauw and Vanhoren 1983), the “Indice biotico esteso” (IBE) in Italy (Ghetti 1986), and many others (Birk and Hering 2002). The Woodiwiss method combines quantitative measures of taxa richness with qualitative information on the sensitivity/tolerance of key indicator taxa.

From a mathematical point of view, the Woodiwiss approach (Trent Biotic Index) does not represent an index calculated with a formula. The biological quality value of a water body is accomplished through the use of the classification table (Table 19.1). The resultant “index” between 1 and 10 is the consequent number at the crossing point of two entrances in the fitting row and column: (1) the vertical row corresponding to the value of number of taxa in the sample and (2) the horizontal column with the fitting key indicator taxa. The biological quality value (1–10) can be transformed into quality class through conversion tables that may vary in different countries, river types, ecoregions, etc. Table 19.1 presents the original table from Woodiwiss (1964). For defining the number of taxa in the sample, i.e., the vertical entrance into the table, the determination level is given for each class or order, e.g., Ephemeroptera and Plecoptera are counted on genus level, Trichoptera and Diptera on family level. The sum of all genera and families in a sample reflects the number of taxa.

The basis of most currently used biotic systems is the Biological Monitoring Working Party system (BMWP) set up by the British Department of the Environment and recommended as biological classification system for national river pollution surveys (Armitage et al. 1983; Hawkes 1997). The BMWP sums up the tolerance scores of all macroinvertebrate families in the sample. Like the saprobic index, the BMWP is based on grouping benthic macroinvertebrates into categories depending on their response to organic pollution. Stoneflies or mayflies, for instance, indicate the cleanest waters and are given a tolerance score of 10. The lowest score (1) is allocated to Oligochaeta, which is regarded to be the most tolerant to pollution.

**Table 19.1** Classification table for deriving the original Trent Biotic Index (Woodiwiss 1964)

Key indicator group	Diversity of fauna	Total number of groups present				
		0–1	2–5	6–10	11–15	16+
Plecoptera nymphs present	More than one species present	–	7	8	9	10
	Only one species present	–	6	7	8	9
Ephemeroptera nymphs present	More than one species present <sup>a</sup>	–	6	7	8	9
	Only one species present <sup>a</sup>	–	5	6	7	8
Trichoptera larvae present	More than one species present <sup>b</sup>	–	5	6	7	8
	Only one species present <sup>b</sup>	4	4	5	6	7
<i>Gammarus</i> present	All above species absent	3	4	5	6	7
<i>Asellus</i> present	All above species absent	2	3	4	5	6
Tubificid worms and/or red Chironomid larvae present	All above species absent	1	2	3	4	–
All above types absent	Some organisms such as <i>Eristalis tenax</i> not requiring oxygen may be present	0	1	2	–	–

<sup>a</sup>*Baetis rhodani* excluded<sup>b</sup>*Baetis rhodani* is counted in this section for the purpose of classification**Table 19.2** Original BMWP score table

Families	Scores
Siphonuridae, Heptageniidae, Leptophlebiidae, Ephemerellidae, Potamanthidae, Ephemeridae, Taeniopterygidae, Leuctridae, Capniidae, Perlodidae, Perlidae, Chloroperlidae, Aphelocheridae, Phryganeidae, Molannidae, Beraeidae, Odontoceridae, Leptoceridae, Goeridae, Lepidostomatidae, Brachycentridae, Sericostomatidae	10
Astacidae, Lestidae, Agriidae, Gomphidae, Cordulegastridae, Aeshnidae, Corduliidae, Libellulidae	8
Caenidae, Nemouridae, Rhyacophilidae, Polycentropodidae, Limnephilidae	7
Neritidae, Viviparidae, Ancylidae, Hydroptilidae, Unionidae, Corophiidae, Gammaridae, Platycnemididae, Coenagrionidae	6
Mesoveliidae, Hydrometridae, Gerridae, Nepidae, Naucoridae, Notonectidae, Pleidae, Corixidae, Haliplidae, Hygrobiidae, Dytiscidae, Gyrinidae, Hydrophilidae, Clambidae, Helodidae, Dryopidae, Elmidae, Chrysomelidae, Curculionidae, Hydropsychidae, Tipulidae, Simuliidae, Planariidae, Dendrocoelida	5
Baetidae, Sialidae, Piscicolidae	4
Valvatidae, Hydrobiidae, Lymnaeidae, Physidae, Planorbidae, Sphaeriidae, Glossiphoniidae, Hirudidae, Erpobdellidae, Asellidae	3
Chironomidae	2
Oligochaeta (whole class)	1

The ASPT (Average Score per Taxon) equals the average of the tolerance scores of all macroinvertebrate families found and thus ranges from 1 to 10 (Table 19.2). The main difference between both indices is that the BMWP system represents the indicative value of taxa diversity while the ASPT does not depend on the family richness. Formerly used as a single or double metric method, nowadays, multi-metric approaches often include national adaptations of the BMWP as a core metric.

The original BMWP method “works” at the family level. Methods developed more recently outside Europe are resolved at higher taxonomic resolutions (Table 19.3), such as the genus and species levels (e.g., HKHbios (Asia), Ofenböck et al. 2010b; ETHbios (Ethiopia), Aschalew and Moog 2015)

**Table 19.3** Taxa and sensitivity scores of benthic macroinvertebrates used in ETHbios calculations (Aschalew and Moog 2015)

Common name	Taxon	Score
Stone flies	Perlidae ( <i>Neoperla</i> sp.)	10
Caddis flies	Lepidostomatidae, Philopotamidae	10
Beetles	Scirtidae	10
Mayflies	Baetidae > 2 spp., <i>Acanthiop</i> sp., Heptageniidae ( <i>Afronurus</i> sp.), Leptophlebiidae	9
Caddis flies	Hydropsychidae > 2 spp.	9
Mayflies	Tricorythidae	8
Caddis flies	Leptoceridae, Ecnomidae	8
Beetles	Psephenidae, <i>Stenelmis</i> sp., <i>Microdinodes</i> sp.	8
Water mites	Hydracarina	8
Crabs	Potamidae	7
Dragonflies, damselflies	Aeshnidae, Lestidae	7
Beetles	Elmidae	7
Crane-flies	Tipulidae	7
Mollusca	<i>Pisidium</i> sp.	7
	Limpets	6
Mayflies	Baetidae with 2 spp., Caenidae	6
Caddis flies	Hydropsychidae with 2 spp.	6
Dragonflies	Gomphidae	6
Water bugs	Naucoridae	6
Horse-flies	Tabanidae	6
Caddis flies	Hydropsychidae with 1 sp.	5
Dragonflies	Coenagrionidae, Libellulidae	5
Water striders	Mesoveliidae, Veliidae, Gerridae	5
Beetles	Hydrophilidae, Dytiscidae, Gyrinidae, Haliplidae	5
Flies	Ceratopogonidae excl. Bezzia-Gr.	5
Mayflies	Baetidae with 1 sp.	4
Water bugs	Corixidae, Pleidae	4
	Belostomatidae, Notonectidae, Nepidae	3
Leeches	Hirudinea	3
Snails	Physidae, <i>Bulimus</i> sp.	3
Midges and Flies	Bezzia-group	3
	Musidae, Chironomidae with predominantly Tanytarsini and Tanypodinae	2
	Psychodidae, Ephydriidae, Culicidae, Red Chironomidae, <i>Chironomus</i> sp., Syrphidae	1
Worms	Oligochaeta	1

Many methodological textbooks mention diversity indices with respect to biotic indices, e.g., Hawkes (1982), Washington (1984), Metcalfe (1989), Johnson et al. (1993), and Resh and Jackson (1993). The diversity approach uses species richness (mostly measured as the total number of taxa), abundance (measured as the number of individuals of each taxon), and evenness (the degree to which each taxon is equally represented) as components of community structure. Unstressed communities are said to be characterized by high diversity (taxa richness) and even distribution of individuals among species. Although a variety of diversity indices exists, there is no normative procedure that can be used for river quality evaluation solely (Kohmann and Schmedtje 1986). In any case, diversity indices are often used in combination with other metrics as a component of multi-metric indices.

## 19.5 The Multivariate Approach

### **Sophisticated, but Laborious in Development**

Multivariate or model-based procedures are predictive systems that assess the deviation between the observed aquatic community and reference conditions predicted from environmental parameters, (e.g., reference condition approach). Models are developed to explain the composition and variability in the aquatic communities among reference sites. The models include a range of environmental parameters. Based on multivariate procedures, the model then predicts what biota should be present at an undisturbed “target” site or river type with a given set of environmental attributes. A study site can be considered in a “very good” or “reference condition” if the aquatic community found at the test site is similar to the predicted one. A study site is considered disturbed if the benthic community observed at the test site is different from the prediction.

Three prerequisites are necessary to successfully apply a multivariate prediction system:

1. A sound knowledge of the species inventory and composition, as well as the spatial and seasonal distribution of the target biota under reference conditions
2. A clear understanding of the criteria that define reference conditions
3. Models that reliably predict the biota for a particular site or river type given the natural variability of environmental conditions

The first remarkable predictive bioassessment tool was RIVPACS, the “River Invertebrate Prediction and Classification System,” developed in the United Kingdom (Wright et al. 1989; Wright et al. 2000; RIVPACS 2005). In the mid-1990s, the Benthic Assessment of Sediment (BEAST) was developed in Canada (Reynoldson et al. 1995). Based on the mathematic principles of RIVPACS, comparable systems have been created in some other countries. To assess the biological health of Australian rivers, the AUSRIVAS (Australian River Assessment Scheme) was developed under the National River Health Program (NRHP) by the federal government in 1994 (AUSRIVAS 2005). The Australian scheme is distinguished by several differences:

the major habitats are sampled and modeled separately, and different models are used for different bioregions over Australia (Simpson and Norris 2000). Since 2001 the PERLA system is in operation in the Czech Republic (Kokes et al. 2006).

Verdonschot (1990) described macrofaunal site groups (cenotypes) in surface waters in the Netherlands, which are recognized on the basis of environmental variables and the abundance of taxa. These cenotypes were described as groups of taxa lumped together based on their limited internal variation. They are distinguished not by zones of overlap in their tolerances or occurrence, since no clear boundaries were provided, but only by a recognizable centroid. The cenotypes are mutually related in terms of key factors, which represent major ecological processes. The cenotypes and their mutual relationships form a web that offers an ecological basis for the daily practice of water and nature management. The web allows the development of water quality objectives, provides a tool to monitor and assess, indicates targets, and guides the management and restoration of water bodies (Verdonschot and Nijboer 2000).

The European Fish Index (EFI) was the first pan-continental model-based index developed for assessing the ecological status of European rivers (Pont et al. 2006). The EFI employs ten metrics describing conditions of fish assemblage regarding feeding, migration, habitat and spawning preferences and tolerance to anthropogenic stress. Site-specific reference conditions are predicted using multiple regression models. An updated version (EFI+) also considers fish length and river-type-specific responses in trout and cyprinid rivers (EFI+ Consortium 2009).

## 19.6 The Multi-metric Approach

### Simple, but Virtuous

The multi-metric approach is currently the most common method among the sophisticated procedures of river status assessment. In the late 1970s limnologists came face to face with the fact that mechanisms of environmental degradation are usually complex, and their combined effects are not easy to measure. Estimates of the biotic integrity of a water body may be the best tool to assess the effect of multiple stressors in aquatic environments. Biotic integrity was defined as “the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region” (Frey 1977; Karr and Dudley 1981). To communicate biological information in a meaningful way, Jim Karr developed the Index of Biotic Integrity (IBI) in the early 1980s and published one of the most cited papers “Assessment of biotic integrity using fish communities” (Karr 1981). The IBI has proven to be very adaptable (Karr and Chu 1999; Simon and Lyons 1995), and quite soon, it has subsequently been adapted for use throughout many states of the United States, and later in many countries of all other continents. Initially developed to monitor fish, the multi-metric approach became extended for aquatic macroinvertebrates, terrestrial macroinvertebrates, macrophytes, algae, wetlands, riparian zones, large rivers, lakes, reservoirs, estuaries, and brackish water ecosystems.



The much-noted experiences of the EU-funded AQEM and STAR projects show that the multi-metric approach is a valuable procedure for bridging the gap between the methodologies and the needs for evaluating the ecological status of water bodies (Hering et al. 2004, 2006; Furse et al. 2006). The multi-metric approach fits quite well to the methodological demands of the European Water Framework Directive, since it attempts to provide an integrated analysis of the biological community of a site by deriving a variety of quantifiable biological characteristics (*metrics*) and knowledge of a site's fauna (Karr and Chu 1999).

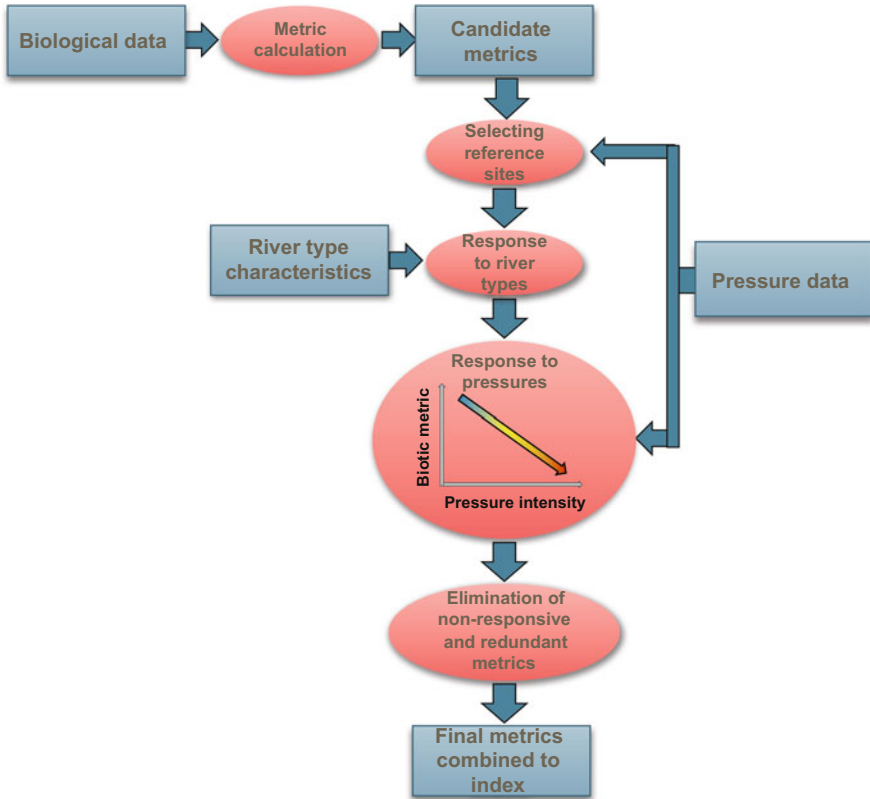
Within a multi-metric index, each single metric is related predictably and reasonably to specific impacts caused by environmental alterations. For example, while the proportion of different feeding types is suited to assess the trophic integrity of an ecosystem, saprobic or acid indices provide a measure to directly assess the impact of certain pollutants and acidification. Thus, the multi-metric index considers multiple impacts and combines individual metrics (e.g., saprobic indices, diversity indices, feeding type composition, current preferences, etc.) into a nondimensional index, which can be used to assess a site's overall condition. By combining different categories of metrics (e.g., taxa richness, diversity measures, proportion of sensitive and tolerant species, trophic structure) reflecting different environmental conditions and aspects of the community, the multi-metric assessment is regarded as a more reliable tool than assessment methods based on single metrics.

The following metric types can be distinguished:

- Composition/abundance metrics. Metrics giving the relative proportion of a taxon or taxonomic group with respect to total abundance. Abundance (or biomass) of a taxon or taxonomic group and/or total abundance (or biomass).
- Richness/diversity metrics. Metrics giving the number of species, genera, or higher taxa within a certain taxonomical entity, including the total number of taxa or diversity indices.
- Sensitivity/tolerance metrics. Metrics related to taxa known to respond sensitively or tolerantly to a stressor or a single aspect of the stressor, either using presence/absence or abundance information.
- Functional metrics. Metrics addressing the ecological function of taxa (other than their sensitivity to stress), such as feeding types/guilds, habitat and flow velocity preferences, ecosystem type preferences, life cycle parameters, and biometric parameters. They can be based on taxa or abundance.

The procedure of data analysis during the development of a multi-metric index typically involves the following steps (Fig. 19.1):

- Selection of the most suitable form of a multi-metric index
- Metric calculation and selection
  - Selection of candidate metrics
  - Exclusion of numerically unsuitable metrics
  - Definition of a stressor gradient
  - Correlation of stressor gradients and metrics
  - Selection of core metrics



**Fig. 19.1** Key steps in developing multi-metric indices

- Distribution of core metrics within the metric types and exclusion of redundant metrics
- Definition of upper and lower anchors and scaling
- Generation of a multi-metric index
  - Development of a general multi-metric index
  - Development of a stressor-specific multi-metric index
- Setting class boundaries
- Interpretation of results

Depending on purpose, ecosystem type, organism group, and available data multi-metric indices may be designed differently. In many cases, a general assessment reliably reflecting the integrity of an ecosystem is sufficient. In other cases, more specific data on the causes of deterioration is required. Thus, we distinguish two main forms of multi-metric index: (1) the general approach and (2) the stressor-specific approach (e.g., Ofenböck et al. 2004). Stressor-specific multi-metric indices

can only be derived if data reflecting different specific stress types and environmental gradients are available and the autecology of the targeted organism group is well known.

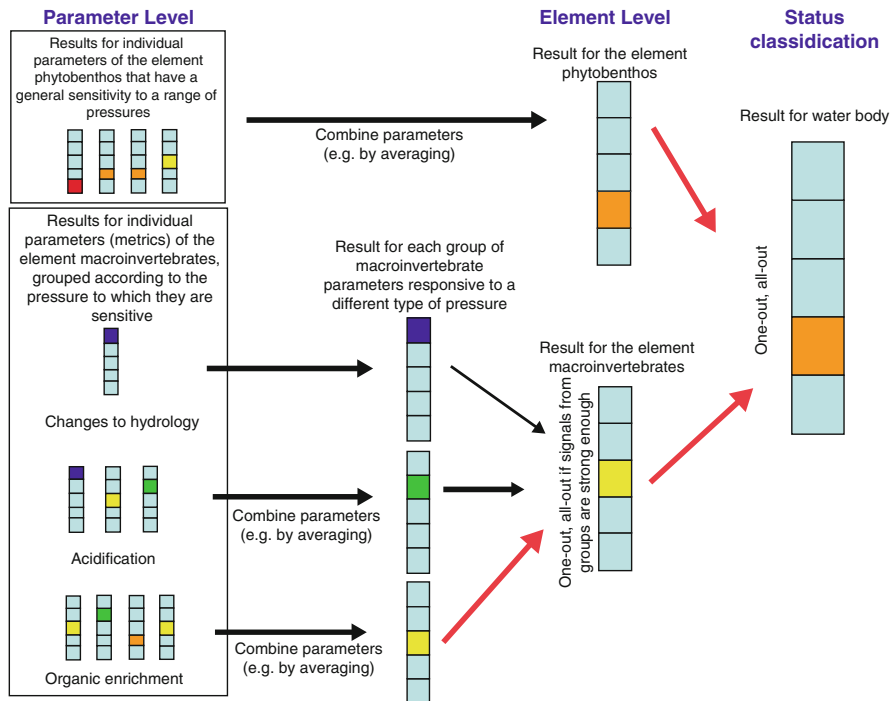
The multi-metric index concept has proven to be very adaptable (Karr and Chu 1999), and many of the same metrics have been used successfully throughout different regions of the world in a variety of stream types (Simon and Lyons 1995). Metrics such as species richness (the total number of taxa) or the EPT approach (number or individuals or % share of Ephemeroptera, Plecoptera, and Trichoptera taxa) are common to most benthic invertebrate-based multi-metric indices.

## 19.7 Integrative Assessment Systems

The most sophisticated evaluation approaches are based on the use of a wide range of organisms that allows an integrated assessment of rivers. In the United States, the Rapid Bioassessment Protocols (RBPs) use biological indicators to infer data about running water quality. RBPs were introduced on a national level in the mid-to-late 1980s (Barbour et al. 1999). There are three main types of RBPs for streams—fish surveys, periphyton surveys, and macroinvertebrate surveys—each with detailed method descriptions. The macroinvertebrate survey is most commonly used, because it requires reasonable expertise or equipment. The EPA encourages the use of RBPs because they provide quick and valid results while being cost effective, time efficient, and minimally invasive (Barbour et al. 1999).

In the southern part of Africa, the SASS (Southern African Scoring System) is seen suitable for the assessment of the ecological integrity of river ecosystems (Dallas 1995, 2007; Dickens and Graham 2002). The South African Assessment Scheme (SAFRASS) protocols use three biotic indicator groups (diatoms, macroinvertebrates, and macrophytes) that respond to changes in river conditions (Lowe et al. 2013).

Since 2000 the Water Framework Directive (WFD) provides a common legal framework for water management in the European Union. The major aim of the WFD is to achieve good ecological status of all European waters (lakes, rivers, and groundwater bodies, transitional, and coastal waters) by 2027 at the latest. Based on annexes II and V of this directive, the EU member states use an integrated system to evaluate the “ecological status” of rivers based on various environmental and biotic features, the so-called quality elements (QE): water chemistry, hydro-morphology, algae, macrophytes, phytoplankton, benthic invertebrates, and fish. The classification scheme for the ecological status of water bodies includes five status classes: (1) very good; (2) good; (3) moderate; (4) poor; and (5) bad. Based on the assessment results of the single QEs, the worst assessment result for a BQE determines the overall assessment result (the “one-out-all-out” principle, Fig. 19.2).



**Fig. 19.2** The one-out-all-out principle of the EU Water Framework Directive (*WFD CIS Guidance Document No. 13: Overall Approach to the Classification of Ecological Status and Ecological Potential*)

## 19.8 Indicator Groups

Various organisms have been used in the assessment of the water quality and ecological integrity of aquatic ecosystems, including bacteria, protozoans, algae, macrophytes, benthic invertebrates, fish, and birds (Roux et al. 1993; Barbour et al. 1999; Bryce et al. 2002). The most frequently used groups are benthic invertebrates, algae, macrophytes, and fish. Current integrative methodologies, such as the US EPA bioassessment protocols, the European Water Framework Directive, or the South African SAFRASS approach, make use of more than one indicator group to evaluate the ecological quality of a water body. To avoid redundant information and thus unnecessary costs, those groups are used to indicate effects of specific stressors on the environment in the most effective way. Algae are perfect indicators to describe the effects of nutrients and eutrophication. The benefit of macrophyte bioindication is to document the effects of long-term nutrient aspects and hydro-morphological impairments. Benthic invertebrates are ideal indicators of (organic) pollution and hydro-morphological deficits at the micro-habitat scale. Fish are supreme indicators to study the effects of hydro-morphological deficits on the meso-habitat and reach scale, including lateral and longitudinal connectivity up to the basin scale.

### ***19.8.1 Periphyton (Contributed by Peter Pfister, ARGE Limnologie, Innsbruck, Austria)***

#### **Tiny, Quick, and Beautiful**

Algae and cyanobacteria are valuable indicators of environmental conditions in running and standing water bodies. As primary producers, periphyton acts as an important foundation of food webs in river ecosystems (Li et al. 2010). Because the assemblages usually attach to substrate, their growing and prospering can respond directly and sensitively to many kinds of physical, chemical, and biological variation occurring in the river reach, including temperature, nutrient levels, current regimes and grazing, etc. Algae are omnipresent in all types of water bodies, and species-rich communities can be found from pristine spring brooks to the effluents of wastewater treatment plants. In Central European running waters, Hürlimann and Niederhauser (2007) recorded densities between  $10^3$  and  $10^6$  individuals/cm<sup>2</sup> on stony surfaces. Their cosmopolitan character and worldwide distribution predestine them as ideal group for nationwide applicable assessment systems. There is a rich knowledge about their ecological requirements, tolerances, and preferences compared to other indicators (Arzet 1987; Hürlimann and Niederhauser 2007; Oemke and Burton 1986; Coring et al. 1999; Rott et al. 1997, 1999; Schmedtje et al. 1998; Tümpling and Friedrich 1999). Rapid reproduction rates and very short life cycles allow algae to react quickly to environmental change. Therefore, periphyton can be expected to reflect short-term impacts and sudden changes in the environment.

From a methodological point of view, the diatoms have a lot of practical advantages: they are comparatively easy to be identified in any stage of their life, one doesn't need a permit to take samples, and the storage of samples or preparations mounted on slides is cheap. On the other hand (which also might be a methodological benefit as it reduces the statistical "noise"), most algae show only little dependency from physical or hydro-morphological factors like flow velocity, substrate type, stream modification, residual flows, interruption of the continuum, and others.

### ***19.8.2 Macrophytes (Contributed by Karin Pall, Systema GmbH, Vienna, Austria)***

#### **Habitat and Food in One**

Macrophytes as autotrophic organisms, first of all, are highly sensitive to nutrient enrichment. This also applies to phytoplankton or phytobenthos. However, two important differences distinguish macrophytes from the latter: their reaction to changes in the trophic state as well as the conclusions which can be drawn from the assemblages found. In principle, all these groups of organisms respond to changes in the trophic level with changes in species spectrum and abundance, though phytoplankton and phytobenthos react much more quickly than macrophytes. Therefore, the former can serve as excellent short-term indicators for rapid detection of

changes in the trophic condition of lakes or rivers. However, repeated investigations are necessary to derive reliable results.

In contrast to phytoplankton and phytobenthos assemblages, macrophyte communities do not show a sudden reaction to changes of the trophic level. They integrate the prevailing conditions over a longer time period. The analysis of the species composition and other features of the macrophyte vegetation thus is a particularly well-suited tool for monitoring long-term trends in trophic conditions. Even from a unique mapping, sound and temporally integrated information about the nutrient conditions in lakes or rivers can be derived. For this reason, the use of macrophytes as indicator organisms for nutrient enrichment already has a long tradition. Usually, as help for water protection institutes, the focus was on the exact localization of organic or nutrient pollution sources along lakeshores or river courses.

However, macrophytes do not solely reflect trophic conditions. They also respond very sensitively to other environmental impacts. In lakes they respond specifically to changes of the hydrological regime (alteration of the natural water level fluctuations) or hydrodynamic conditions (e.g., changes of the wave frequency and intensity by motorboats or navigation). In rivers they have proven to be excellent indicators for changes in the flow regime (e.g., potamalisation or rhithralisation). Furthermore, the specific composition of the macrophyte community is a pronounced reflection of the structural conditions found along the shores and in the water body of lakes and rivers, such as e.g., substrate diversity and dynamic or the degree of embankments. Therefore, last but not least, the macrophyte vegetation can serve as indicator for the structural alterations of the shoreline and the quality of the water-land-linkage.

In particular, two properties make macrophytes highly valuable indicators. On the one hand, it is their longevity. They remain at the same sites over several vegetation periods and thus can integrate the site conditions over a considerably longer time period than other quality elements as e.g. phytoplankton and phytobenthos. On the other hand, macrophytes always remain in the same place and are thus not able to avoid pressures and other environmental impacts, e.g., benthic invertebrates or fish. This enables an accurate localization of the sources and the spheres of impact of pressures.

### ***19.8.3 Aquatic Macroinvertebrates***

#### **Tiny but Many: Helpful Creepy-Crawlers**

Benthic macroinvertebrates are the most widely used indicator group for lotic systems. There are several advantages to using benthic macroinvertebrates in bioassessment, because they constitute a substantial proportion of freshwater biodiversity and are critical to ecosystem function. The following list summarizes briefly the advantages of benthic macroinvertebrate bioindication (Danecker 1986; Hellawell 1986; Moog 1988; Metcalfe 1989; Rosenberg and Resh 1992; Metcalfe-Smith 1994; Ollis et al. 2006).

- Benthic macroinvertebrates are widespread and can be found in most aquatic habitats.
- There are a large number (thousands) of species.
- From the point of systematics and phylogeny, they are a highly diverse group, which makes them excellent candidates for studies of changes in biodiversity.
- Different systematic groups of macroinvertebrates have different environmental needs and tolerances to pollution or other kinds of stressors.
- Benthic invertebrates cover a broad range of micro- and meso-habitats, ecotones, biocoenotic regions, trophic position (trophic interaction), etc.
- Macroinvertebrates feed on micro-/mesofauna as well as on algae and are the primary food source for fish. Therefore, an impact on macroinvertebrates impacts the food web and designated uses of the water resource.
- Small order streams often do not support fish but do support rich macroinvertebrate communities.
- Macroinvertebrates are to some extent mobile and can actively select habitats that fulfill their environmental needs.
- On the other hand, benthic invertebrates have limited mobility, and thus they are indicators of local environmental conditions.
- Since benthic invertebrates retain (bioaccumulate) toxic substances, chemical analysis will allow detection in them where levels are undetectable in the water resource.
- A biologist experienced in macroinvertebrate identification will be able to determine relatively quickly whether the environment has been degraded by identifying changes in the benthic community structure.
- Benthic macroinvertebrates have the ideal size to be easily collected and identified.
- Sampling of macroinvertebrates under a rapid assessment protocol is easy, requires few people and minimal equipment, and does not adversely affect other organisms.
- In the industrialized world, there is a good knowledge on identification, procession, and evaluation of benthic invertebrates.

#### ***19.8.4 Fish***

##### **Tasty and Valuable Indicators**

Fish communities respond significantly and predictably to many kinds of anthropogenic disturbances, including eutrophication, acidification, chemical pollution, flow regulation, physical habitat alteration, fragmentation and introduced species (Li et al. 2010). Their sensitivities to the health of surrounding aquatic environments form the basis for using fishes to monitor environmental degradation. Over the last three decades, a variety of fish-based indices have been widely used to assess river quality, and the use of multi-metric indices, inspired by the index of biotic integrity (IBI), has grown rapidly.

- Fishes are present in most surface waters except in cases of stream size and migration restrictions.
- The identification of fishes is relatively easy, and their taxonomy, ecological requirements, and life histories are generally better known than in other species groups.
- Fish presence corresponds strongly to changes in hydrological and environmental flow patterns, while other biological quality elements (e.g., macroinvertebrates) hardly indicate these impacts.
- Fishes have evolved complex migration patterns, making them sensitive to continuum interruptions.
- The longevity of many fish species enables assessments to be sensitive to disturbance over relatively long time scales.
- The natural history and sensitivity to disturbances are well documented for many species, and their responses to environmental stressors are often known.
- Fishes generally occupy high trophic levels and thus integrate conditions of lower trophic levels. In addition, different fish species represent distinct trophic levels: omnivores, herbivores, insectivores, planktivores, and piscivores.
- Fishes occupy a variety of habitats in rivers: benthic, pelagic, rheophilic, limnophilic, etc. Species have specific habitat requirements and thus exhibit predictable responses to human-induced habitat alterations.
- Depressed growth and recruitment are easily assessed and reflect stress.
- Fishes are valuable economic resources and are of public concern. Using fishes as indicators confers an easy and intuitive understanding of cause effect relationships to stakeholders beyond the scientific community.

There is a common agreement that the performance of any biological assessment approach increases with the quality rating of its ecological background (Verdonschot and Moog 2006). Consequentially there was a remarkable increase of taxa lists that associated ecological information with indicator taxa in the last 10–15 years. These taxa lists include functional ecosystem characteristics, species traits, and others more in ecological assessment (see Chap. 20).

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