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Biological Indicator Systems in Floodplains – a Review

key words: bioindication, environmental assessment, biodiversity, RIVA project

Abstract

Based on a literature review, the different approaches to biological indicator systems in floodplains are summarised. Four general categories of bioindication are defined and proposed here: 1. Classification indicators, 2.1 Environmental indicators, 2.2 Biodiversity indicators, 3. Valuation indicators. Furthermore, existing approaches in floodplains are classified according to the four categories. Relevant and widely used approaches in floodplains are explained in more detail. The results of the RIVA project are put into the context of these indication approaches. It is concluded that especially functional assessment approaches using biological traits of the species can be seen as very promising and deserve more attention by conservation biologists and floodplain ecologists.

1. Introduction

The influence of organic pollution on aquatic organisms is so obvious that these observations have made quite a significant contribution to the development of the idea of bioindication. Aristotle is said to have been the first scientist to point out the connection between organic pollution and changes to aquatic biocoenoses (Thienemann, 1959). Then the concept of bioindication arose with work on the saprobic index by Kolkwitz and Marsson at the beginning of the 20th century (Cairns and Pratt, 1993). Aquatic organisms have not only been important for devising indicator systems for organic pollution but also for the development of numerous other indicator systems for the condition of rivers, lakes, and canals and for evaluating human impacts on aquatic systems (Knoben *et al.*, 1995; Statzner *et al.*, 2001). Today indication is widely used to describe and evaluate environmental conditions and to assess the success of environmental policies with easily recordable indicators.

Indicators for complex ecosystems such as floodplains, which are determined by parameters and processes that are difficult to measure directly (e.g. the frequency and duration of inundation), are of special importance (e.g. Foeckler and Bohle, 1991; Schubert, 1991; McGeoch, 1998; Statzner *et al.*, 2001; Dziock *et al.*, in press).

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In floodplains a bias towards aquatic compartments exists regarding the compartments for which attempts have been made to develop or apply indicator systems. Also, only a few attempts have been made to review indication systems that are relevant for floodplain systems and most of them are limited to the aquatic compartment and/or focus on the implementation of specific policies (e.g. Knoben *et al.*, 1995; Statzner *et al.*, 2001; Dziock *et al.*, in press; www.eurolimpacs.ucl.ac.uk).

The object of this article is to review already established indicator systems in the context of different bioindication concepts. For each indication system, we briefly discuss what is indicated, whether different indicators are equally suitable, whether standards have been developed, and whether the transferability of the system has been tested. We conclude with summarising major gaps that exist in the development of indicator systems for floodplains as a whole.

2. Concepts of Bioindication

Bioindication refers to the use of animals and plants as instruments for assessing past, current, or future conditions or processes. The particular advantage of bioindication is that animals and plants must cope with partly changing or fluctuating environmental conditions for a fairly long period and so integrate in the course of this fairly long period. Individual measurements of chemical parameters are only snapshots and may produce other figures shortly beforehand or afterwards. Bioindicators are accordingly species or groups of species that provide information about the long-term quality of environmental changes and fluctuations (McGeoch, 1998). A precise definition of the indication goals and the environmental factors to be indicated is of elementary importance, but is not carried out in many instances (Lindenmayer, 1999). Several authors addressed the issue of categorising bioindication goals (e.g. Schubert, 1991; McGeoch, 1998; Zehlius-Eckert, 1998; Lindenmayer *et al.,* 2000). Most schemes can be converted into one another. Here we merge the concepts of McGeoch (1998) and Zehlius-Eckert (1998), classifying bioindication systems according to their purpose and the entity they indicate (see Fig. 1):

1. **Classification indicators** indicate object characteristics that serve to distinguish between classes. They allow objects to be assigned to classes and the result of the indication is also shown in an appropriate fashion. The HGMU (Hydrogeomorphological Unit)-approach (Maltby *et al.*, 1996) and the typology approaches of Castella (1987), Richardot-Coulet *et al.* (1987), and Foeckler (1991) are examples for a classification indication. They can be used to assign wetlands or waterbodies to categories based on abiotic parameters and the organisms living in them. There are transitions between classification and environmental indicators (2.1), especially if the classification builds on specific environmental factors. Therefore, the use of these approaches for classification and environmental bioindication can present two sides of the same coin.

2. **Status indicators** indicate object characteristics that serve to describe objects. The result of the indication is in most cases shown quantitatively (duration of inundation: 2–3 weeks per year) or relatively (longer flooding in test area 1 than in test area 2). Indicator values serve as an example. They reflect the realised optima of species along a gradient of an environmental variable. Examples are to be found mainly in plant ecology (Diekmann, 2003), but also increasingly for other organisms (e.g. diatoms: Denys, 2004; molluscs and carabids: Follner and Henle, 2006). Status indicators can be environmental indicators or biodiversity indicators.

2.1. **Environmental indicators** indicate an abiotic or biotic state. In most cases a very precise definition of the factor to be indicated is possible. Usually, it is based on known causal connections, for example the impact of hydrodynamics on organisms in floodplains (e.g. Statzner and Higler, 1986). And so bioindicators can be related to quantifiable abiotic parameters, such as the duration or frequency of flooding, oxygen content of the water, soil moisture, pH value of the soil, or intensity of use. A quantitative connection of this nature has so far been made, especially among the lotic fauna (saprobic system: FRIEDRICH, 1990; MAUCH *et al.*, 1990), aquatic molluscs (RICHARDOT *et al.*, 1987; FOECKLER 1990, 1991; FOECKLER *et al.*, 1991) and vegetation (indicator value system as described by LONDO, 1975 and ELLENBERG *et al.*, 1992), whereas in the case of other organisms, especially terrestrial ones, we have thus far much less knowledge of quantifiable connections with abiotic factors (but see for example FALKNER *et al.*, 2001 and SPEIGHT, 2005).

2.2. **Biodiversity indicators**: a group of species indicates the presence of members of one or more other taxonomic groups. It is in almost all cases impossible to record all species or measure all the biodiversity present in a given area. Therefore, attempts have been made to find species groups that are indicative of the overall biodiversity or the overall species richness (e.g. FAVILA and HALFFTER, 1997; LAWTON *et al.*, 1998; EEA, 2004). This approach requires that the different groups either depend on each other or they are driven by a similar set of environmental factors such as disturbance regime (MCINTYRE *et al.*, 1999), edaphic factors (CODY, 1986), or the biogeographic history (SPECTOR, 2002). Therefore, it is not surprising that tests or applications of this concept have led to inconsistent results (e.g. HILDEBRANDT *et al.*, 2005b). Regarding floodplains, the success of such an indication of biodiversity is highly dependent on the region, water type, and the associated habitat diversity (NIJBOER *et al.*, 2005).

3. **Valuation or target indicators** indicate object characteristics that serve to put targets into concrete terms, such as target figures, and then valuate objects relative to these targets. The indication result is a value appraisal in categories such as "worth protecting" or "protection target achieved or missed". The evaluation particularly requires standardisation, for which there have been widely formulated proposals (PLACHTER, 1994; PLACHTER *et al.*, 2002).

3. Established Indicator Systems for Floodplain Compartments

There are numerous methodological approaches to indication, which are used for floodplains as a whole or specific floodplain compartments (e.g. KNOBEN *et al.*, 1995) but only a few of them have been systematically expanded into standardised and tested indicator systems. Approaches have been suggested for all four categories of indication goals, apparently most frequently for the purpose of environmental indication, but often value systems were added later and the initial environmental indication system was then used as well for value indication. Below we discuss bioindicator systems already established or currently being tested that are important for floodplains. These are ordered according to their indicative purpose (Fig. 1, Table 1).

3.1. Classification Indicators

These indicators serve to delimit spatial units (such as habitats or hydro-geomorphological units) from others mostly by the presence of the indicator in the one spatial unit or the lack of it in the other spatial unit. Indicators can be species (in plant sociology, e.g. characteristic or differential species), but also certain abiotic parameters (morphology, hydrological factors, soil parameters etc.). INNIS *et al.* (2000) provide an overview of the types of indicators used. Prominent examples for the latter are the Hydrogeomorphic classification for wetlands in North America (HGM, BRINSON, 1993, 1996) which forms the basis for the HGMU approach in Europe (MALTBY *et al.*, 2006) or hydrological indices (review in OLDEN

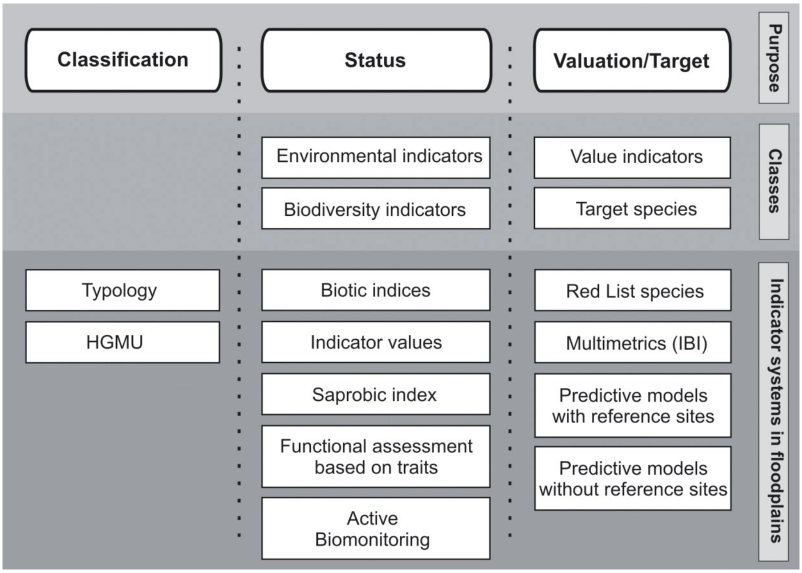

Figure 1. Selection of existing indicator systems in floodplains and a classification according to the purpose of the indication.

and POFF, 2003). The approaches used in biological indication are subsumed here under the typology approach. However, the different classification systems used in floodplains differ considerably in their criteria used for the classification.

3.1.1. Typology Indicators

These indicators are used for the characterisation and description of biotypes, their transitions and a selected set of their functions. Classic examples include the running water organisms in the continuum from the source, upper, middle, and lower reaches to the estuary (ILLIES, 1961; WRIGHT *et al*., 1984; BRAUKMANN, 1995) and the classification of running water by the landscape areas of high and low mountain ranges and lowland (WRIGHT *et al*., 1988; BRAUKMANN, 1994). There are similar approaches for various floodplain biotypes using plant and animal species as classification indicators (e.g. ZAHLHEIMER, 1979; CASTELLA, 1987; GERKEN, 1988; FOECKLER, 1990; 1991; CASTELLA *et al*., 1994; HÜGIN and HENRICHFREISE, 1992; FOECKLER *et al*., 1995b, DEVILLERS *et al*., 1991; RIECKEN *et al*., 2003; DRACHENFELS 2004).

The procedure for working out the classification indicators is often based on the construction of "functional describers" sensu CASTELLA and AMOROS (1988). In these cases certain species or their combinations are identified, whose main purpose is a classification, e.g. a specific phytosociological association, but which at the same time can be used as environmental indicators to describe the functional situation of the biotopes they live in (e.g. groundwater influence, water dynamics, semi-aquatic etc.). For instance brown trout (*Salmo trutta*) not only define a whole biocoenosis and stream region connected to this species but also indicate its habitat characteristics and function (oxygen rich, cool flowing water).

Table 1. Selection of major bioindicator systems classified according to their purpose.

Class	Name of indicator system/approach	What is indicated?	Indicators used	Standards developed?	Transferability tested?	Major references
1. Classification indicators	Classification of wetlands and deepwater habitats of the United States	hierarchical grouping of wetland habitats indicating ecosystem integrity	vegetation	yes	yes, extensively	COWARDIN *et al.*, 1979
	Typology approach	function by allocating sites to classes	vegetation, molluscs, fish, macrozoobenthos	yes	yes	CASTELLA *et al.*, 1994; FOECKLER *et al.*, 1995a,b
	HGMU	wetland functions	HGMU (a landscape unit of uniform geomorphology and hydrological regime, vegetation)	yes, partly	in progress	MALTBY *et al.*, 2006
2. Status indicators						
2.1 Environmental indicators	Biotic indices, (excluding indices of biotic integrity, these are grouped under value indicators)	local environmental variables	macrozoobenthos, diatoms, fish	partly	yes, but many similar methods in different countries	KNOBEN *et al.*, 1995
	Saprobic index	saproby	macroinvertebrates	yes	yes	ROLAUFFS *et al.*, 2004
	Indicator values of higher plants	local environmental variables	plants	yes	yes, extensively	ELLENBERG *et al.*, 1992; ERTSEN *et al.*, 1998; DIEKMANN, 2003
	Indicator values of other organisms	local environmental variables	diatoms, molluscs, carabid beetles	no	yes	DENYS, 1994; FOLLNER and HENLE, 2006
	Functional assessment based on species traits	habitat properties, environmental variables, human impact	multiple traits of invertebrates and other organisms	not yet	yes	STATZNER *et al.*, 2001; BADY *et al.*, 2005

Table 1. (continued)

Class	Name of indicator system/approach	What is indicated?	Indicators used	Standards developed?	Transferability tested?	Major references
	Functional metrics of fish assemblages	local environmental variables → after comparison with reference condition: ecosystem integrity	fish	novel index	on a European scale (5252 sites!)	Pont *et al.*, 2006
	Active biomonitoring	chemical pollution	fish, crustaceans, and others	yes	yes	Schirmer *et al.*, 2002
2.2 Biodiversity indicators	Biodiversity indication	biodiversity of a taxon group, overall biodiversity	presence or abundance of taxa	no	no	EEA, 2004; UNEP, 2001
3. Valuation indicators	Red List species	extinction risk/value of the community	all taxa with legal protection status or where Red Lists exist	yes	yes	IUCN, 2003
	Multimetrics (indices for biological integrity, IBI)	quality/biological integrity	macrozoobenthos, amphibians	yes, IBI, Karr and Chu, 2000	different indices necessary for each country (Hering *et al.*, 2004)	Karr and Chu, 2000; Hering *et al.*, 2004, Chovanec *et al.*, 2005
	Predictive models with reference sites	human impact/river health	macroinvertebrates	yes	yes	RIVPACS: Hawkins *et al.*, 2000; Clarke *et al.*, 2003
	Predictive models without reference sites	human impact/quality	hoverflies (Diptera, Syrphidae), macro-invertebrates	no	yes, partly	Speight and Castella, 2001; Chessman and Royal, 2004

Another approach using mostly abiotic data is the derivation of hydrogeomorphological units (HGMUs) (MALTBY *et al.*, 1996). An HGMU is a landscape unit of uniform geomorphology and hydrological regime. Vegetation units, soil type, hydrology, and geomorphology are used as indicators for HGMUs and their ecosystem functions. These floodplain functions (retention, groundwater discharge etc.) can then be assessed by interrogating a database containing all the data available for the HGMU (also socio-economic criteria). The approach can in principle be used for all floodplain compartments and is only limited by the degree of differentiation and the availability of data (MALTBY *et al.*, 2006). A special advantage of this procedure is the possibility of predicting the impact of human intervention in the floodplain system (SCHOLZ *et al.*, 2004; JANSEN *et al.*, 2005).

3.2. Status Indicators

These indicators aim to describe current, past, or future states or processes. A large subgroup is formed by those indication systems that are based on calculating biotic indices, such as species indicator values, the saprobic index, and functional assessment based on traits (Table 1). A further group is status indicators that are used for active biomonitoring of pollutants. All in all, so many biotic and diversity indices are described that below we can only present a subjective selection and must refer to more extensive literature (METCALFE, 1989; CAIRNS and PRATT, 1993; KNOBEN *et al.*, 1995).

Biotic indices discussed in this section have no normative component. Thus, they belong to environmental indicators. We do not include multimetric indices here because in most cases their calculation is dependent on the definition of some reference status as can be seen from the definitions provided by KARR and DUDLEY (1981) and NORRIS and HAWKINS (2000) for the aquatic zone and ANDREASEN *et al.* (2001) for a terrestrial index of biotic integrity. Therefore, we discuss them under value indicators.

3.2.1. Environmental Indicators

Attempts at methodological development and spatial transferability of systems for environmental indication in floodplains have been based in various approaches. On the one hand, expert knowledge and literature have been combined to evaluate the connections between a group of species and their environment and/or the significance of habitat for the existence of species (e.g. VERNEAUX *et al.*, 1982; FOECKLER, 1990, 1991). On the other hand, the evaluation of indication characteristics of a species or habitat is frequently based on empirical assessments and a generalisation of ecological knowledge. Classic examples are indication by higher plants (e.g. LONDO, 1975; ELLENBERG *et al.*, 1992) and the saprobic index for describing the pollution of bodies of water by organic substances (FRIEDRICH, 1990; MAUCH *et al.*, 1990).

Another approach consists in trying to decide upon groups of organisms or abiotic parameters suitable for indicative purposes by comparing different areas or various sectors within an area (e.g. DISNEY, 1986; KÖPPEL *et al.*, 1994; MALTBY *et al.*, 1996). Here too, observations are frequently dealt with empirically, but the use of recent advances in multivariate statistical methods, such as logistic regression models and generalised additive models (MCCULLAGH and NELDER 1989; PEARCE and FERRIER, 2000), co-inertia analyses (CASTELLA *et al.,* 1994), including fuzzy coded variables (CASTELLA and SPEIGHT, 1996), can provide substantial progress in knowledge regarding the connections between abiotic and biotic parameters, from which indication systems can be deduced (cf. CASTELLA *et al.*, 1994; HENLE *et al.*, 2005b). The RIVA project has extended this approach by using a carefully worked out experimental plan to localise the sampling spots (RINK *et al.*, 2000; HENLE *et al.*, 2006).

3.2.1.1. *Biotic Indices*

Biotic indices are applied to assess biological properties of mainly running waters, in most cases they are based on macroinvertebrate-communities. Biotic indices have been used to measure various types of environmental stress, organic pollution, acid waters, etc (KNOBEN *et al.*, 1995). Several dozens of different biotic indices have been developed, one of the oldest being the saprobic index (KOLKWITZ and MARSSON, 1908) and the Trent Biotic Index (WOODIWISS, 1964). In the following, we discuss some of the most widely used indicator systems based on biotic indices.

3.2.1.2. *Saprobic Index*

The saprobic index for running water is one of the oldest indication systems. Initial approaches stem from KOLKWITZ and MARSSON (1902, 1908, 1909). It was developed for use in flowing water and indicates the saprobity, i.e. the organic pollution of a body of water. It is the most widely used indicator system relevant for running waters in Central Europe (MARGREITER-KOWNACKA *et al.*, 1984; ROLAUFFS *et al.*, 2004) and has even received an industrial standard in the Czech republic (CSN 75 7716, 75 7221) and Germany (DIN 38410), where it has been the standard method for assessing running water since the mid-1970's. In the current revision of DIN 38410 of 2004 it is described as a measure for biological valuation based on the benthal fauna when implementing the European Water Framework Directive (EU-WFD 2000, Annex 5). Thus, in addition to being used as an environmental indication system it now serves also as a valuation indication system. This required that environmental values are attached to the indices. Since the natural saprobity differs for different types of water bodies, a one-to-one correspondence between the index and the environmental value does not exist. Although originally designed for running waters, efforts are underway to expand the approach to lakes. The Lake Biotic Index, for example, is related to trophic potential and bottom dissolved oxygen, and organic matter in the sediment (VERNEAUX *et al.*, 2004). These authors also discuss the potential of the approach for lake diagnosis.

3.2.1.3. *Indicator Values of Higher Plants*

The use of indicator values of higher plants (e.g. ELLENBERG, 1974; LONDO, 1975; ELLENBERG *et al.*, 1992) is, like the saprobic index, a long established, successful, and widely implemented indication system. The indicator values relate to site conditions, for which plants are particularly suitable because of their lack of mobility. The indicator value systems of ELLENBERG *et al.* (1992) and LONDO (1975) have an empirical basis, i.e. they are essentially heuristic. Initially, they were not developed for the specific dynamic conditions of floodplains. However, statistical procedures in the RIVA project showed that the indicator value system of ELLENBERG *et al.* (1992) and the moisture types of LONDO (1975) are also suitable for describing soil moisture under the special conditions of floodplains (AMARELL and KLOTZ, in press). This is in concordance with the results of SCHAFFERS and SYKORA (2000). Both indicator systems can be used for the terrestrial area of floodplains to indicate soil moisture, soil reaction, and light conditions. Their advantage (and at the same time their disadvantage) lies in the fact that they indicate site conditions in an integrated form, but are not suitable for indicating specific hydrological parameters such as the average level of the groundwater table (semi-)quantitatively. For general discussions of the advantages and limits of these plant indicator systems, we refer to BÖCKER *et al.* (1983), WIEGLEB (1986), KOWARIK and SEIDLING (1989), ERTSEN *et al.* (1998) and DIEKMANN (2003).

3.2.1.4. *Indicator Values of Other Organisms*

The calculation of species indicator values for organisms other than plants has been rarely conducted. One example is the study of DENYS (2004) for diatoms. He observed a rather poor performance of diatom species to adequately indicate measured environmental variables, such as pH, salinity, saprobity, etc. He argues that the usefulness of diatom indicator values for these environmental variables is limited.

A bioindicator system based on indicator values calculated on the basis of weighted averages of measurements of two water parameters (duration of inundation, groundwater depth) is presented by FOLLNER and HENLE (2006). Indicator groups are plants, carabid beetles, and molluscs. To our knowledge, this is the first system to successfully indicate factors related to the dynamics of water levels on a quantitative basis. Performance was tested and found to be accurate in terms of spatial and temporal transferability of the system for all three groups.

3.2.1.5. *Functional Assessment Based on Species Traits*

This conceptual approach has its roots in the habitat templet concept of SOUTHWOOD (1977, 1996) developed further by TOWNSEND and HILDREW (1994), and POFF (1997). Habitat is seen as a templet that shapes the life history traits of the species living in that habitat. Numerous empirical studies have shown the validity of this approach (TOWNSEND and HILDREW, 1994; STATZNER *et al*., 1997; TOWNSEND *et al*., 1997). Subsequently, it has been used to develop biomonitoring tools that use general biological traits of organisms that indicate ecological functions (DOLÉDEC *et al.*, 1999; USSEGLIO-POLATERA *et al.*, 2000; STATZNER *et al*., 2001; GAYRAUD *et al*., 2003; BADY *et al*., 2005). Functional assessment approaches use the biological characteristics of the entire species pool for an indication and link biological traits of the observed species community with ecological properties of the habitat. The occurrence or lack of certain ecological functions, such as flooding or drying out, is determined by assessing the incidence of species with similar habitat requirements (FOECKLER, 1990, 1991; CASTELLA *et al*., 1994; CASTELLA and SPEIGHT, 1996). Functional assessment can indicate both qualitatively and quantitatively a particular ecological function. It can be employed for different scales such as small sectors of landscapes or for entire regions (e.g. STATZNER *et al*., 2001; GAYRAUD *et al*., 2003). A simplified approach has been presented as the ITC (index of trophic completeness; PAVLUK *et al.*, 2000). The index uses the presence or absence of 12 trophic groups among the macroinvertebrate fauna as an indicator of trophic completeness of the river under study. This is functional assessment focussed on the biological trait “feeding type”. Recently, PONT *et al.* (2006) presented an approach combining functional assessment with a predictive modelling approach (see below). By means of using functional metrics and not species alone, they managed to develop a novel fish biotic index transferable between catchments at the European scale.

Even in freshwater habitats, where most of the research on monitoring programs has so far been concentrated (eg. STATZNER *et al*., 2001; GAYRAUD *et al*., 2003), the use of biological traits is only in its experimental phase (BADY *et al*., 2005). In the terrestrial compartments of the floodplain, functional assessment using biological traits is even less common (e.g. EU project Functional Analysis of European Wetland Systems FAEWE, CASTELLA *et al*., 1994; MALTBY *et al*., 1996). The RIVA project (HENLE *et al*., 2006a) is based on these approaches and advanced our ability to indicate site conditions in floodplain grassland using functional assessment based on vascular plants (AMARELL and KLOTZ, in press), molluscs (FOECKLER *et al*., 2006), and syrphids (DZIOCK, 2006). This involved the application of multivariate statistical methods, such as co-inertia or RLQ (DOLÉDEC *et al*., 1996), to be able to relate multiple trait variables to multiple environmental variables. The RIVA results showed the high potential of that approach, although it became clear that the necessary life history data for a specific taxonomic group exists only for very few organism groups (e.g. vascular plants, shelled Gastropoda, hoverflies (Syrphidae): KLOTZ *et al.*, 2002; FALKNER *et al.*, 2001; SPEIGHT *et al.*, 2004). Even in these published databases, some of the species-level data have been extrapolated from congenerics, leading to a clustering of congenerics, when classified into ecological groups based on their life history.

Phylogeny is also a confounding factor and has to be taken into account. This is because species descend from their ancestors in a hierarchical fashion, and related species tend to resemble each other in their biological characteristics. Therefore in comparative studies, analysis methods have to control for phylogeny (HARVEY and PAGEL, 1991; DESDEVIDES *et al.*, 2003). Up to the present, no published study is known to us that has related habitat to multiple life history characteristics and involved a control for phylogeny. Studies on this topic are currently underway. Among these insights at the species level, the RIVA studies (DZIOCK, 2006; FOECKLER *et al.*, 2006; GERISCH *et al.*, 2006; AMARELL and KLOTZ, in press) have shown that a successful approach for running waters, the classification of species into functional groups at the generic level, usually does not work within wetland and terrestrial floodplain habitats.

3.2.1.6. *Active Biomonitoring*

For these studies standardised test organisms are inserted into the environment to test their reaction to certain environmental factors. Such studies are frequently used to distinguish between the effect of different types of pollution. They serve inter alia to determine toxicity, bio-accumulation, or mutagenicity (KNOBEN *et al.*, 1995; DAYEH *et al.*, 2002; NEUMANN *et al.*, 2003a, b). We cannot provide a survey at this point in view of the wide dissemination of such test procedures and the abundance of literature (SCHIRMER *et al.*, 2002).

3.2.2. Biodiversity Indicators

EEA (2004) gives an overview over existing water-related biodiversity indicators, i.e. bioindicators that can be used to measure biodiversity of a taxon or as a whole. There seems to be little work done on this topic in floodplains. Indicators used are, for example, the diversity of fish families, macrophyte species composition, or benthic macroinvertebrates (UNEP, 2001). We are far from a standard procedure for evaluating these approaches, but some authors point out that the success of such an indication of biodiversity is highly dependent on the region, water type, and the associated habitat diversity (NIJBOER *et al.*, 2005).

3.3. Valuation Indicators

3.3.1. Red List Species

Red lists are one of the most frequently used instruments to indicate the ecological value of parts of landscapes, but alternative valuation systems do exist. For example, SMARDON (1983) discusses the suitability of visual-cultural effects on observers as valuation indicators for floodplains.

The number of endangered red list species found in a particular area is defined as a measure for the nature preservation value of the area. This criterion has been applied in many floodplain areas, e.g. the Danube (FOECKLER, 1990) and the Salzach (FOECKLER *et al.*, 1995a). Ideally, red lists contain a relative estimate of the likelihood of extinction of a species (IUCN, 2003). Internationally, there have been several attempts to develop objective criteria for red listing species (e.g. Mace and Lande, 1991; Cogger *et al.*, 1993). Nevertheless, most of the criteria are difficult to apply to the majority of species, especially those that are directly linked to quantified extinction risks. Additionally, the fact that red lists fulfil social and political functions as well as that of an assessment instrument, i.e. they incorporate social and political value systems, limits their "ecological objectivity" (Binot *et al.*, 1998).

Arguments solely based on levels of endangerment are inadequate from a technical point of view, whether or not species are worth protecting depends on higher-ranking biogeographical, ecological, economic, and ethical grounds (Plachter, 1994; Fritzlar and Westhus, 2001). Thus misjudgements are possible, if red lists are used as the sole value indicator system. De Nooij *et al.* (2004, 2005) compared ecological and policy-based biodiversity assessments in a floodplain, i.e. site valuation based on species richness and the number of red list and legally protected species (European Habitats Directive etc.), respectively. As results differed in some sites and for some taxonomic groups, they argue that an ecological approach and a policy-based approach (like red lists or legal protection) yield complementary information on the value of floodplains.

3.3.2. Target Species

Target species are species that serve the formulation and testing of concrete objectives of nature conservation (abbreviated in accordance with Zehlius-Eckert, 1998). This brings them under the category of Valuation/target indication formulated by us above. They can be used not only to operationalize objectives in nature conservation but also more concretely as parameters by which the success of nature protection measures and landscape conservation can be measured (Bouwma *et al.*, 2003; Rosenthal, 2003). Further details can be found e.g. in Hildebrandt (2001) and Zehlius-Eckert (1998).

3.3.3. Multimetrics (Indices of Biological Integrity, IBI)

Multimetrics (so called to distinguish them from biotic indices) have been promoted in an effort to reduce complexity and present data in a form that will be easy for non-experts to understand (Norris and Hawkins, 2000). In North America, the multimetric approach attempts to quantify the concept of biological integrity by giving values to several different biological attributes (e.g. taxa composition) and comparing them to the values given by a defined reference status, which serves as a target. Then, an overall index of biological integrity (IBI) can be calculated from the values of the individual indicators. This index is said to reflect the concept of river health (Norris and Thoms, 1999; Norris and Hawkins, 2000). The use of indicators for biological integrity in North America has been reviewed recently (Adamus *et al.*, 2001). In the course of the implementation of the EU Water Framework Directive, indicator systems similar to the IBIs used in North America have been set up for European use. One example is the AQEM system developed for 28 European stream types using a database with nearly 10 000 invertebrate taxa (Hering *et al.*, 2002, 2004). The AQEM approach uses quality classes ranging from 5 (high quality) to 1 (bad quality). A formal statistical approach is presented by Dodkins *et al.* (2005) for river macrophytes. Multivariate ecological quality assessment metrics are calculated from a set of reference sites. Then it is possible to compare these metrics for the reference and the test sites. The method can be used to distinguish non-impacted from impacted sites (Dodkins *et al.*, 2005).

Although the IBI approach has been widely used in stream assessment, developments of similar indices for the terrestrial compartments are rare (Bradford *et al.*, 1998; Andreasen *et al.*, 2001). CHOVANEC *et al.* (2005) propose a new index very similar to the IBI that also makes use of organisms in the aquatic terrestrial transition zone (dragonflies, amphibians, molluscs).

3.3.4. Predictive Models with Reference Sites

These models with reference sites quantify river health as the degree to which a site supports the biota that would be expected to occur there in the absence of alteration by humans (the reference or target condition) (HAWKINS *et al.*, 2000; NORRIS and HAWKINS, 2000). These indicator systems measure the deviation of the ecological quality of running waters or floodplains from quality targets of environmental policies, one of the oldest being the RIVPACS system ("River Invertebrate Prediction and Classification System") that has been used in the UK for rivers and brooks since 1978 (MOSS *et al.*, 1987; WRIGHT *et al.*, 1989; CLARKE *et al.*, 1996). A comparative set of reference flow sections serves to determine the optimal status. Recording and balancing out numerous environmental variables such as geographic width and length, flow characteristics, width of flow, substrate categories, and water forge data enables the existing macroinvertebrate fauna to be forecast. This is compared with the fauna recorded on-site. Conclusions are drawn with regard to the current ecological status of the water compared with the reference status from the differences between the actual colonisation and the one expected for the water without human impacts. Indices serve to quantify the comparison between the expected and the actually existing fauna. The RIVPACS system uses further information on the species, the so-called BMWP (Biological Monitoring Working Party) score, which is the sum of the values assigned to the species in the (English) saprobic system. The quotient from the BMWP score of the fauna predicted and the BMWP score of the fauna actually detected then forms part of the classification for the sections of water under investigation. The basic idea is that a quotient with a similar value indicates a similar water quality, irrespective of the type of water body (CLARKE *et al.*, 2003). The relatively simple calculation for this index makes the procedure striking and easily communicable to the general public, even if the prediction model on which it is based uses highly complex multivariate procedures such as discriminant analyses or cluster procedures (HAWKINS *et al.*, 2000; TER BRAAK *et al.*, 2003). Similar systems with the same philosophy as RIVPACS but with partly differing calculation algorithms and statistical methods are employed in North America (BEAST, Benthic Assessment of Sediment, REYNOLDSON *et al.*, 1995) and Australia (AUSRIVAS, NORRIS and NORRIS, 1995; TURAK *et al.*, 1999; for a review see LINKE *et al.*, 2005). Predictive models and multimetrics (IBI) have many things in common, a detailed critique, advantages and disadvantages for the two methods can be found in NORRIS and HAWKINS (2000).

3.3.5. Predictive Models without Reference Sites

Most of the predictive model approaches use sets of reference sites to create a benchmark for the biological assessment of the river condition. Unfortunately, human modification of river systems is now so widespread that finding appropriate non-impacted sites can be difficult or impossible. Therefore, it is desirable to be able to make predictions without having to define a reference status. For the terrestrial and semi-aquatic floodplain area, a bioindication system has been developed based on European hoverflies (Diptera, Syrphidae), with habitat preference data being used for the prediction (SPEIGHT and CASTELLA, 2001; SPEIGHT and GOOD, 2001; SPEIGHT, 2005). Again, the comparison between the suite of predicted taxa and the taxa actually observed at an assessment site provides an indication of human impact. Evaluation software for this already exists (SPEIGHT *et al.*, 2001), but there are not yet any established indices or evaluation schemes. Work on this topic is progressing and the results are extremely promising (e.g. SPEIGHT and CASTELLA, 2001; SPEIGHT, 2005). Similar data is available for western European snails (FALKNER *et al.*, 2001) that could facilitate the development of similar approaches for molluscs.

For running waters, CHESSMAN and ROYAL (2004) have independently developed a similar bioindication system for macroinvertebrates using environmental filters (annual water temperature range, flow regime, river bed composition) and the regional taxa pool for the prediction of macroinvertebrate communities. An assessment using this method showed substantial correlation of the proportion of observed and predicted taxa with several independent measures of human impact (CHESSMAN and ROYAL, 2004).

4. Discussion and Conclusion

In this review paper, bioindicator systems in floodplains have been presented and classified according to their indication purpose into four groups: classification indicators, environmental indicators, biodiversity indicators, and valuation indicators. Most of the bioindicator systems available for floodplains belong to the environmental indicator and the valuation indicator categories. Furthermore, it has become obvious that only a very small fraction of the systems presented in this review has been specifically developed for floodplains. The overwhelming majority is designed for the use in running waters and can, at most, be converted for the use in the aquatic terrestrial transition zone (ATTZ sensu JUNK, 1989).

Examples for indicator systems designed specifically in and for floodplains are the predictive models without reference sites by CASTELLA and SPEIGHT (1996) and SPEIGHT and CASTELLA (2001), the HGMU-approach by MALTBY *et al.* (2006), and the RIVA indicator system using plants, carabid beetles, and molluscs to indicate the duration of inundation and groundwater depth (FOLLNER and HENLE, 2006). There seems to be a considerable lack of true biodiversity indicators for floodplains, as in the two recent review compilations (UNEP, 2001; EEA, 2004) not a single study of biodiversity indication is listed that was specifically designed for floodplains or floodplain organisms. Here we see an urgent need to test and develop methods specifically for the ATTZ.

There has been some debate concerning the use of multimetric approach (biological integrity indices) versus the predictive models approach. Both approaches define a reference condition based on reference sites, presumed to have low or absent human alteration. NORRIS and HAWKINS (2000) summarise the Pros and Cons for these two approaches and conclude – in response to KARR and CHU (2000), that the predictive model approach offers many advantages over the multimetric approach for assessing river health. Another conclusion of these authors is that persuasive arguments are not everything, but what is urgently needed is a means of objectively testing the performance of different approaches. This holds true not only for the two approaches mentioned above, but also for the other approaches. FOLLNER and HENLE (2006) have exemplified a rigorous testing procedure for their bioindicator system. They systematically compared the performance of different bioindicators in terms of precision and bias when temporally and spatially transferred. We need more of these tests and comparisons to make our proposed indicator approaches more robust and usable for nature conservationists.

DOLÉDEC *et al.* (1999) argue that the U.S. (multimetric approach) and the British approaches (predictive models) for monitoring the biological integrity of lotic ecosystems could be improved by clearly integrating life-history patterns and resistance or resilience potential of the species of a community. We could not agree more. Particularly the functional assessment approach deserves more attention by nature conservation researchers, and can be seen as very promising, as it enables us to identify similar habitat conditions and hence similar ecological strategy types of the organisms living there, even when the observed species pools in two compared habitats differ for biogeographical reasons. This approach is especially suitable for highly dynamic and highly vulnerable ecosystems like floodplains. Thus functional assessment based on traits could serve to assess the degree of succession/recovery or degradation after disturbances (e.g. hydraulic engineering or natural flood events like the extreme Elbe flood of 2002). It would be very interesting to see how the floodplain community changes in functional terms after such events.

PONT *et al*. (2006) seem to have taken up the functional assessment challenge and manage to combine the best of both worlds: they present a predictive model approach, but use life history traits of fishes instead of species data as functional metrics. Thus they predict functional attributes of fish species on a given site by the local environmental variables (i.e. mean annual air temperature, geological type, flow regime etc.). Afterwards, the deviation of the observed functional attributes of the actual fish fauna from the predicted values of the functional attributes is calculated for each trait and transformed into a quality scale giving the desired output of a biological quality index. A similar approach is presented by HEINO (2005) who explores functional diversity and functional evenness measures in relation to environmental variables. He discusses also applications of these functional diversity measures for management and conservation. These two appraoches present major steps in the development of indicator systems.

The results of the trait analyses presented in this special issue (DZIOCK, 2006; FOECKLER *et al*., 2006; GERISCH *et al*., 2006) could be used for forecasting systems like that of PONT *et al*. (2006) using the data of FALKNER *et al*. (2001) and SPEIGHT and CASTELLA (2001).

The species environment relationships from the RIVA project (FOECKLER *et al*., 2006; HENLE *et al*., 2006b) are already partly implemented in forecasting models used by German Federal Waterways authorities (FUCHS *et al*., 2003) to predict ecological changes under alternative management options. By modifying the models such that they predict traits as functional attributes, not species, these modelling tools could see a geographically much wider application.

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