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Preface

This report has been prepared by R.A.E. Knoben and C. Roos (Witteveen + Bos Consulting Engineers, The Netherlands), in close cooperation with M.C.M. van Oirschot (RIZA, The Netherlands). The guidance-committee on this report comprised of M. Adriaanse, E.C.L. Marteijn, P.J.M. Latour and J.G. Timmerman (RIZA, The Netherlands). The report has been reviewed by the international experts: G.A. Friedrich (LWA, Germany), P. Logan (NRA, United Kingdom), E. Nusch (Ruhrverband, Germany), N. de Pauw (University of Gent, Belgium) and H. Soszka (Poland).

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1. Introduction

1.1 General

The Convention on the Protection and Use of Transboundary Watercourses and International Lakes (hereinafter referred to as the Convention) was drawn up under the auspices the Economic Commission for Europe and adopted at Helsinki on 17 March 1992. The Convention was signed by 25 countries and by the European Community before the period of signature closed on 18 September 1992. It will enter into force 90 days after the date of deposit of the sixteenth instrument of ratification, acceptance, approval or accession. By the time of writing of this report, thirteen countries and the European Community had deposited their relevant instruments of ratification with the United Nations Secretary-General.

To comply with the obligations under the Helsinki Convention, the Parties will, inter alia, have to set emission limits for discharges of hazardous substances from point sources based on the best available technology. In addition, they will have to apply at least biological treatment or equivalent processes to municipal waste water. They shall also issue authorizations for the discharge of waste water and monitor compliance. Moreover, they have to adopt water quality criteria and define water quality objectives. To reduce the input of nutrients and hazardous substances from diffuse sources, in particular from agriculture, they shall develop and implement best environmental practices. Furthermore, environmental impact assessment procedures and the ecosystem approach shall be used to prevent any adverse impact on transboundary waters.

Consequently, the Helsinki Convention addresses such issues as monitoring, assessment, warning and alarm systems, and exchange and presentation of information. For example, the Parties bordering the same transboundary waters will have to set up joint or coordinated systems for monitoring and assessment of the conditions of transboundary waters, and set up coordinated or joint communication, warning and alarm systems. The clear objective of monitoring and assessment systems such as the Helsinki Convention is to prove that changes in the conditions of transboundary waters caused by human activity do not lead to significant adverse effects on flora and fauna, human health and safety, soil, air climate, landscape and historic monuments or other physical structures or the interaction among these factors.

The establishment of a system to furnish proof that these objectives are met is a challenging task. Moreover, monitoring compliance with the provisions of the Helsinki Convention demands reliable information on waters and factors influencing water quality and quantity. There is, for instance, a need for information related to in-stream quality, such as conditions of waters (water quantity and quality), aquatic and riparian flora and fauna, and sediment. Information related to extreme conditions in waters, caused by accidents, floods, drought or ice cover, is also needed. Emission sources also have to be monitored to obtain information on the concentration of pollutants in effluents, and to carry out pollution-load assessments.

Consequently, information on monitoring of surface waters and significant emission sources in catchment areas of transboundary waters is required. This includes information on the legal basis of emission monitoring, selection of variables, selection of sampling sites and frequencies and documentation and reporting of the results (both to authorities and to the public at large). Information on monitoring for early warning purposes, including biological warning systems, is required as well.

Following the adoption of the Convention, the Senior Advisers to ECE Governments on Environmental and Water Problems (now known as the ECE Committee on Environmental Policy) entrusted its Working Party on Water Problems with the implementation of the Convention, pending its entry into force. To implement the work plan, the Working Party has set up several task forces and groups of rapporteurs. The topics addressed are:

1. point sources;
2. diffuse sources;
3. legal and administrative aspects;
4. sustainable water management;
5. monitoring and assessment.

The present report has been prepared within the context of the Task Force on monitoring and assessment, which was led by the Netherlands.

This Task Force has been charged with the preparation of draft guidelines to ECE Governments on monitoring and assessment. During the first meeting of the Task Force, a phased approach towards this goal has been approved. During the first phase, the focus will be on ‘running-water’ transboundary water courses (i.e. rivers, streams, canals), while in later phases, the focus will be on lakes, estuaries and groundwaters.

The present report is one in a series of 5 background documents to be used for the drafting of guidelines on monitoring and assessment of running-water transboundary water courses. These reports deal with the following themes:

1. inventory of transboundary rivers and international lakes in Europe;
2. inventory of current monitoring and assessment practices in UN/ECE countries;
3. preparation of draft guidelines for biological assessment of rivers;
4. preparation of draft guidelines for quality assurance;
5. inventory of State of the Art practices in monitoring and assessment.

The present report is the result of the activities under item number 3: Biological Assessment of Rivers.

1.2 Study objectives

The objectives for this desk-study are:
- preparation of a literature review on the international state-of-the-art biological assessment and presentation methods;
- evaluation of routine biological monitoring and assessment methods in UN/ECE countries and comparison of these current practices with state-of-the-art;
- formulation of recommendations for short-term and long-term harmonisation efforts.

In order to meet the second objective RIZA has performed an enquiry amongst ECE-countries (especially those represented in the Task Force) by means of a questionnaire. The questionnaire contains a number of questions about biological monitoring activities. Chapter 4 will discuss the reported results.

1.3 Scope and restrictions

Biological assessment can be defined as the systematic use of biological responses to evaluate changes in the environment with the intent to use this information in a quality control program (Matthews et al., 1982). This definition is often used in a restricted sense in which biological assessment refers to field studies on plankton, macroinvertebrate or fish community in a river to evaluate biological water quality. In this sense, biological assessment is a form of ecosystem monitoring (De Zwart, 1994).

In this report, however, the area of study has been extended from biological assessment in this restricted sense to assessment methods that take more aspects of the riverine ecosystem into consideration, such as habitat quality assessment and ecological assessment. Furthermore, assessment methods that use bioindicators of other biotic groups or apply an experimental setup with organisms, like toxicological methods, are considered in this report as biological assessment methods. Also attention will be given to the future perspective of integrated assessment (De Zwart, 1994). Biological early warning systems (bio-alarm) for discharges of river quality control are however not included.

The study has been limited to assessment methods for watercourses or running waters such as rivers, streams and canals. Methods for standing water bodies, like lakes and reservoirs, have been excluded. A less profound restriction has been applied to the geographical distribution of the application or occurrence of biological assessment methods. Most emphasis has been put on the European continent and more specific the Helsinki countries , but some important methods from other countries are incorporated in the literature review as well.

At the start of this study, it was clear that the number of existing methods was overwhelming. For this reason it was decided to present and discuss categories of methods, illustrated with some examples.

The recommendations presented in chapter 5 of this draft report are only preliminary and result purely from the desk study. It is felt that the step from these preliminary recommendations to transboundary guidelines needs further discussion with participating countries. These discussions should preferably include "technical" as well as legal, political and organizational aspects. It would be very helpful in discussing these matters to recognize both short-term and long-term goals for harmonization. Long-term objectives could be used to coordinate future developments, while the more practical short-term goals facilitate the exchange of relevant data of the watercourses between countries.

2. Watercourses in ecological perspective

2.1 Watercourses as part of riverine ecosystems

A watercourse or river is an open system with a strong directionality and strong interactions with its drainage basin. Four dimensions in environmental relationships between river and surrounding landscape can be distinguished: longitudinal, lateral and vertical gradients and a temporal dimension (figure 2.1; from Ward & Stanford, 1989).

Distinct longitudinal gradients from headwaters to downstream estuary are a result of dynamics in hydrology and morphology, together with spatial differences in geology, relief and soil in the catchment area. Typical examples are (in downstream direction) increasing river bed width and depth, decreasing stream velocity, decreasing substrate grain size and increasing enrichment by nutrients. This longitudinal gradients in abiotic determining factors result in an ecological zonation of communities, both functional and structural, from origin to river mouth, as illustrated in the River Continuum Concept (Vannote *et al.*, 1980).

The transverse or lateral gradient in natural streams and rivers can be appointed in the way the aquatic zone (water body) of a riverine ecosystem is interlinked with the riparian zone (banks, amphibious zone) and the terrestrial zone (floodplains). Abiotic determining factors like erosion and sedimentation patterns and stream velocity differ greatly between streambed, banks and floodplains, inner and outer curves etc.

A third gradient or dimension is the vertical relationship between the river sediment and underlying groundwater system. Finally, a temporal dimension can be considered in the duration of certain natural events like floods and other changes in water level. Moreover, there is a temporal dimension in the time scale of man induced impacts, for example in the way dams prevent migratory fish movements and regulation prevents natural variations in water level (Ward & Stanford, 1989).

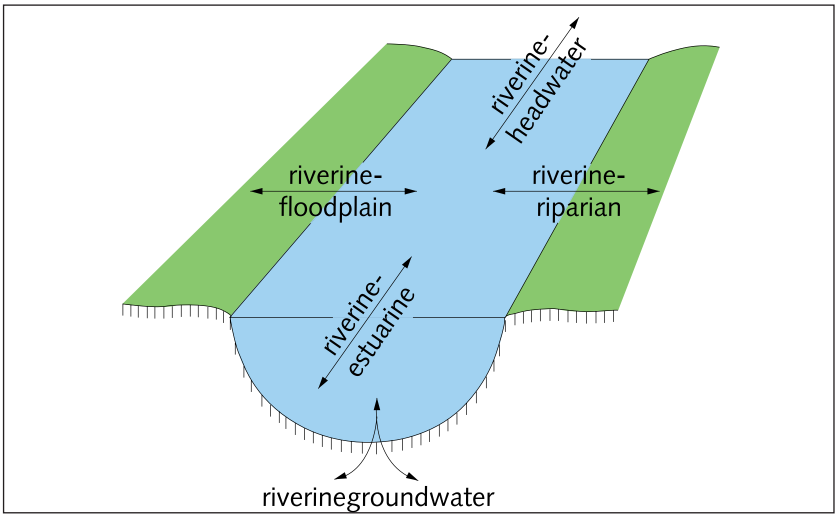
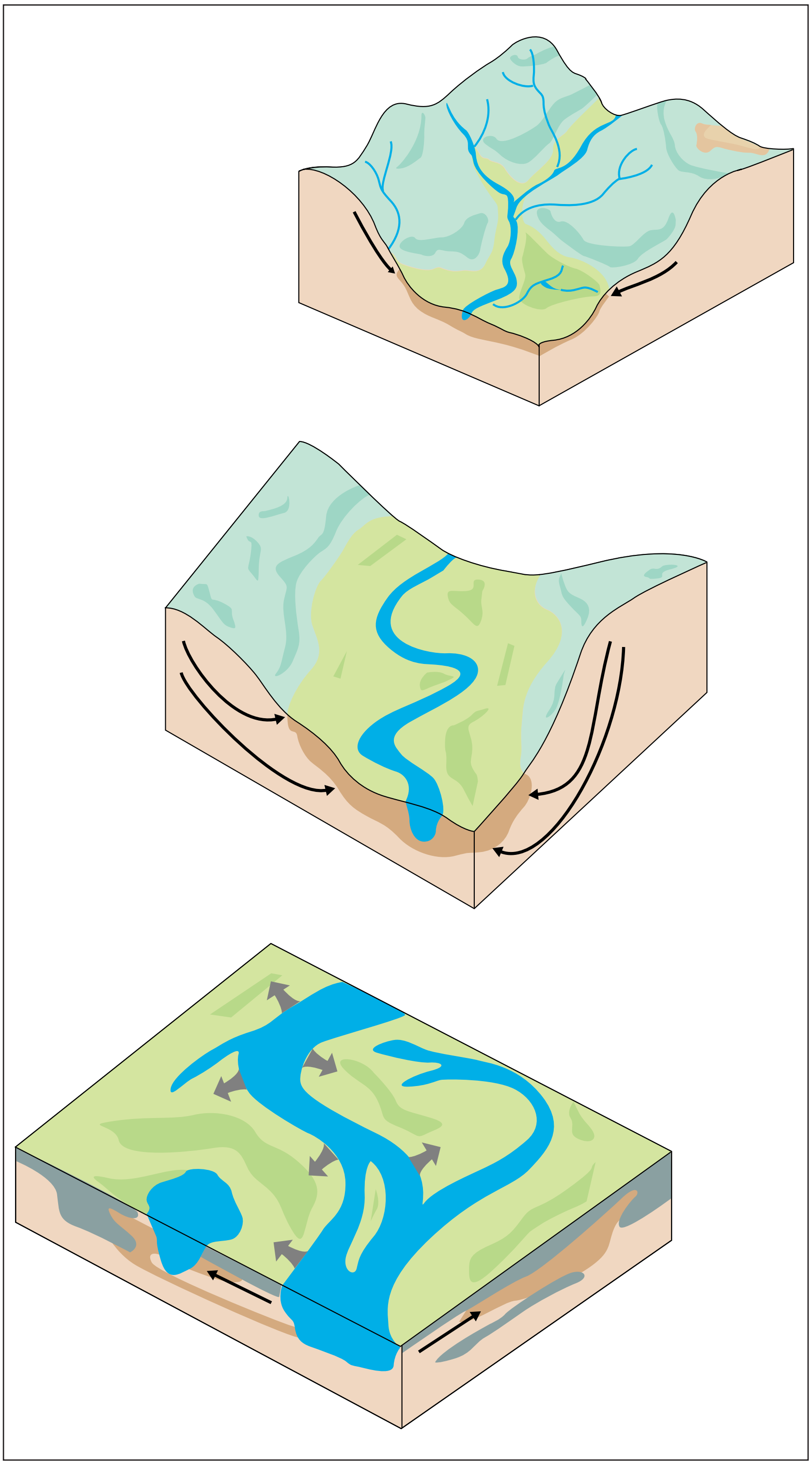


Figure 2.1
Major interactive spatial pathways of reverine ecosystems [from Ward & Stanford, 1989].

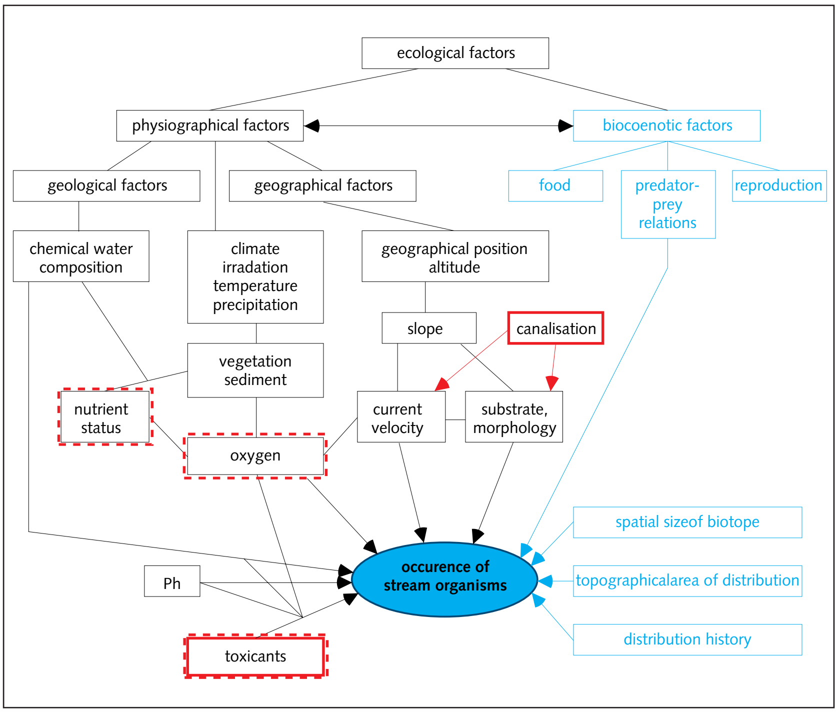
These gradients lead (potentially) to a large variation of habitats in and along natural rivers which in turn result in a large differences in species composition of communities. Several manuals on the ecology of natural rivers as well as impacted streams already date from the seventies (Hynes, 1970; Ward & Stanford, 1979). At that time major emphasis was put on the communities of the aquatic zone or water body only, including their interrelations with abiotic determining factors like current velocity, substrate and chemical water composition.

Figuur 2.2
Ecological relations at landscape level of a river in its environment in three reaches: upper, middle and lower part.



The multiple relationships between both environmental conditions and biological relations in benthic communities within watercourses account for a complex scheme, for example illustrated by Braukmann for small running waters or brooks (figure 2.2; modified from Braukmann, 1987 and De Pauw & Hawkes, 1993).

Figure 2.3
Determining factors in occurrence of benthic organisme in running waters [translated from Braukman, 1987; updated with De Pauw & Hawkes, 1993]. Black= abiotic factors; green = biological factors; dashed red = factors which are in use for water quality criteria; solid purple = unnatural or anthropogenic determinants.



An impression of the predominant relations on a landscape ecology scale between a river and its natural environment is given in figure 2.3 for three reaches of the river. In the upper reach interrelations concern mainly discharge and erosion. In the lower reach, the river can have a major impact on the terrestrial zone, by processes like deposition of suspended solids, supply of foreign species, etc. as a result of floods.

Although the aquatic zone has received most attention last decades, attempts to classify the other riverine ecosystem zones have been described. Rademakers & Wolfert (1994) distinguished 18 coherent types of habitats - called ‘ecotopes’ - varying from floodplain forests and meadows to reed marshes and side-channels. This approach can be useful in ecological rehabilitation of floodplains (IRC, 1992). Of course not all ecotopes will necessarily be present in a specific river; the study demonstrates the variety that can exist under natural circumstances. The habitat variation however forms the conditional matrix for the species diversity and the complexity of the foodweb. Also, habitat diversity determines many natural values like key species/taxa in nature conservation (e.g. plants, amphibians, water birds and mammals). As an example, flowing side-channels along rivers and associated floodplain woodlands highly increase the species diversity (e.g. Barneveld *et al.*, 1993).

In addition to hydromorphological dynamics, other factors determine the actual development of the riverine ecosystem. Due to river pollution, land use dynamics and morphological adjustments like canalization or weirs, impairment of natural ecosystem development occurs. Many disturbing effects are known at the species level as well as at whole community level due to (flow) regulation, acidification, eutrophication, and toxic discharges (e.g. Griffith, 1992).

The actual aquatic community can thus be considered as the integrated biotic response to all existing abiotic and biotic forces. This holistic view on riverine ecosystems has to be made measurable in order to be of practical use in ecological water management. Therefore a number of representative and sensitive parameters have to be selected to monitor and assess the watercourse. Sufficient knowledge of river ecosystem functioning is a prerequisite to the correct selection of representative and sensitive parameters. The use of multivariate statistics is necessary to find which environmental variables account for most variation in the original data.

2.2 Classification of rivers

Assessment of river quality implies the activity of measuring biological or ecological status on a certain (linear or non-linear) scale, which preferably is furnished with clear endpoints. At one side of the yardstick, at the low levels of quality, the assessment endpoint appears to be well defined: "dead water". The other end however, can be considered the status of a part of the river ecosystem under natural conditions or reference state and is far more difficult to define. Classification could help to define natural variety in rivers.

At this point, two major ecological concepts need to be mentioned: the classical concept in which a river is divided into particular zones, and the concept of a water course as a continuum of communities. As a result of the former concept, a classification scheme was proposed on a worldwide scale in 1963 (Illies & Botosaneanu, 1963). Another proposal to establish a macrohabitat based classification on the scale of the European community was presented by Persoone (1979). He distinguished 432 macrohabitats. The River Continuum Concept was introduced in 1980 and regards a river as a continuum of communities that differ both in structure and in function (Vannote *et al.*, 1980). The applicability of this concept to (very) large rivers as well as small rivers is however argued (Sedell *et al.*, 1989; Verdonschot, 1990).

Verdonschot (1990) has reviewed and discussed the advantages and disadvantages of both concepts. He reaches the general conclusion that classification and continuum are not contrary, but rather supplementary concepts. Consensus on this issue can be reached by combining the pragmatic part of classification and the recognition of abstract conceptions with the realism of the multidimensional model of the continuum approach.

At a regional or national level several typological classifications for running waters have been made (e.g. Verdonschot, 1990; Friedrich, 1993; Wright *et al.*, 1993). At present however, no biotypological classification of rivers exists that can provide reference sites and aquatic communities at the scale of the European countries. Furthermore, one has to realise that assessment of whole riverine ecosystems following an ecosystem approach will require a more extensive set of reference data which has to be extended to communities of banks, floodplains etcetera. The availability of an European database of reference sites would be of great interest to integrated river management. An attempt in defining 'ecotopes' for the terrestrial and amphibious zones of river Rhine have been made by Rademakers & Wolfert (1994). This approach needs to be integrated with aquatic community classifications.

2.3 Historical development from physical-chemical to ecological assessment

The assessment of water quality for water management purposes has until now been based on physical, chemical or biological data, or a combination of these. Although the physical-chemical monitoring methods of inland waters are probably the oldest, biological monitoring has a tradition of almost a century or even longer, given the first documented observation that polluted waters contained other faunal species than clean waters (Kolenati, 1848).

Chemical water quality assessment however received the most attention of policy makers and was implemented at a much earlier stage in legislation and standards (water quality objectives) than biological assessment. Some important factors may have been:
- the direct relation with emissions of polluting substances;
- the relative ease to perform and standardise sampling and measurements of 'common' chemicals in river water;
- the straight-forward manner in which water management objectives and quality standards can be expressed in terms of threshold concentrations;
- the manner in which deterioration of water quality of watercourses due to pollution and the subsequent loss of drinking water supply or other functional uses addresses the public interest more directly than loss of biological quality.

Chemical assessment does not provide direct information on the effects of pollution on the biological quality or ecosystem health of the river. To obtain a more complete picture of water quality, the assessment can be extended to biological assessment. A number of important specific features of biological assessment can be mentioned (Metcalfe, 1989):
- biotic communities integrate environmental conditions over a long period of time and require low-frequency sampling whereas chemical analysis offer snapshots of single moments, requiring a high frequency;
- the actual number of substances present in surface waters exceeds the number of measured substances by orders of magnitude (Van Leeuwen, 1995). For many (toxic) substances no analysis methodology is available or environmental concentrations are below detection levels;
- water quality objectives and uses that are related to aesthetic, recreational and ecological dimensions can only be expressed in terms of biological or ecological features and be assessed by biological methods only.

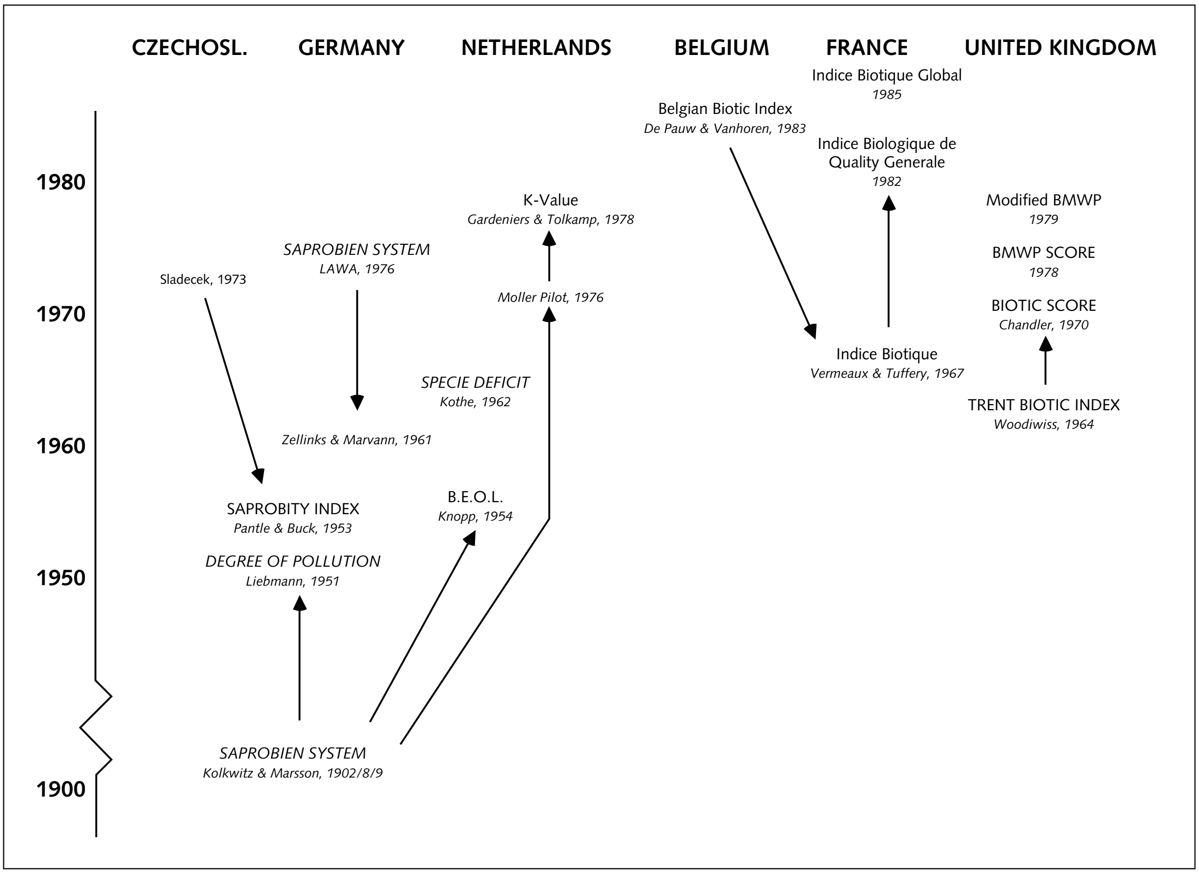
Chemical and biological assessment of water quality can serve different purposes and can consequently be considered complementary rather than mutually exclusive.

The classic saprobic system based on the presence of species developed by Kolkwitz & Marsson and later extended by Liebmann (1962), has provided a scientific and practical method for classifying the impact of organic pollution of running waters by combining chemical and biological aspects (Kolkwitz & Marsson, 1902,1908,1909). The application of the saprobic system has been increased strongly by the possibility of quantifying the results with the aid of the saprobic index S or a modified index including saprobic valencies (Pantle & Buck, 1955; Zelinka and Marvan, 1961). Sládecek offered a comprehensive summary and revision of the development of water quality assessment methods from the biological point of view (Sládecek, 1973). After his publication of an extensive list of water organisms as indicators of saprobity, the saprobic system was and is up till now applied in many European countries (see 3.5 and 4.4).

The assessment of biological water quality by means of macroinvertebrates originates from the United States (Richardson, 1928). In Europe, the first development in the use of benthic communities for water quality assessment, apart from the saprobic system, arose in the United Kingdom and was first presented in the Trent Biotic Index (Woodiwiss, 1964).

Since the late seventies, three rounds of international testing and evaluation of the major biotic indices have been performed in (West-)Germany, the United Kingdom and Italy, initiated and encouraged by the EEC (Tittitzer, 1976; Woodiwiss, 1978; Chesters, 1980; Ghetti & Bonazzi, 1980). A comprehensive description of the historical development and evaluation of biotic, saprobic and diversity index methods based on macroinvertebrates is presented by Metcalfe (1989) (see 3.2 and 3.3). A recent overview of applications in the countries of the European community is given by De Pauw & Hawkes (1993). Figure 2.4 summarizes the essentials of both chronological overviews (modified from Metcalfe (1989) and De Pauw *et al.* (1992) (after Woodiwiss,1980).

Figure 2.4
Chronological development and geographical distribution of biological assessment in some European countries [modified from Metcalfe, 1989 & De Pauw *et al.*, 1992 (after Woodiwiss, 1980)].



During the seventies, the focus of water quality problems shifted from organic load to eutrophication and toxic effects of polluting substances. Recently the interest changed again to the quality of the aquatic ecosystem as a whole, including both the water zone or water body itself and the interlinked system of the aquatic (including water bottom or sediment), riparian and terrestrial zones and the animal and plant communities present there.

This ecosystem approach is being pursued because of the insight into the strong interaction between all relevant abiotic conditions and the biotic response of the aquatic community.
For this reason, attempts have been made to develop ecological assessment methods (Roos *et al.*, 1991; Laane & Lindgaard-Joergensen, 1992; Friedrich *et al.*, 1993; Klapwijk *et al.*, 1995). This development towards integrated assessment can also be recognised in the United States in the assessment of the ecological integrity, in which both the biological condition and the habitat quality are evaluated (Plafkin *et al.*, 1989; Barbour *et al.*, 1992).

2.4 Towards an integrated approach

For a long time, management of watercourses was dedicated to human functional uses and was concerned mainly with hydro-morphological aspects. Relevant hydrodynamic processes such as rainfall, discharge characteristics and inundations were carefully studied. Furthermore morphodynamics were of interest because of the strong relationship of erosion, sediment transport and sedimentation to hydrodynamics. Flood-control, shipping and water supply could be managed sufficiently by monitoring only these variables.

At present, this type of information is insufficient to meet current demands in water quality and water quantity management. Functional uses like drinking water production, fisheries, industry and agriculture often co-exist and all make their respective water quality demands. Developments in aquatic ecology show that rivers have intrinsic ecological “functions” that need to be protected, such as species and habitat diversity, foodweb interrelations and production and mineralisation of organic matter. These functions and uses are in turn related to hydrology, morphodynamics, water quality, etc. For this reason, a more integrated type of water management is obviously needed, addressing the functioning of the aquatic ecosystem as a whole, including its use. As a working-title, “integrated catchment management” could be used for this approach. This approach is also promoted by the Helsinki convention (UN/ECE, 1992).

Adopting this approach will have consequences with respect to monitoring and assessment objectives and activities. Knowledge of ecosystem performance under natural conditions will have to be used to elucidate specific interrelations within the ecosystem, and to define management targets and (possible) bottle-necks. Targets will have to include both ecological targets (which can be seen as an intrinsic functional use) and functional (or use related) targets, related to each other in a logical and coherent way to avoid conflicting management.

In summary, one can conclude that most traditional biological assessment methods, like saprobic or biotic indices, no longer provide a sufficient tool to integrated water management due to their restricted approach to one or few aspects of the water ecosystem. Chapter 3 of this report will discuss in further detail some promising new methods as well as the existing and proven methods.

2.5 Assessment objectives in an integrated approach

State-of-the-art monitoring and assessment of riverine ecosystems should involve both biotic and abiotic variables, both water quantity and quality aspects, both aquatic and floodplain features, and both structural and functional variables. In many cases, functional uses are incorporated as well in quality objectives. An example of a state-of-the-art set of variables is presented in the provisional EU-directive (European Union, 1994) in which ecological quality is determined by the following target variables:

- dissolved oxygen and concentrations of toxic or other harmful substances in water, sediment and biota;
- levels of disease in animal life, including fish, and in plant populations due to anthropogenic influence;
- diversity of invertebrate communities (planktonic and bottom-dwelling) and key species/taxa normally associated with the undisturbed condition of the ecosystem;
- diversity of aquatic plant communities, including key species/taxa normally associated with the undisturbed condition of the ecosystem, and the extent of macrophytic or algal growth due to elevated nutrient levels of anthropogenic origin;
- the diversity of the fish population and key species/taxa normally associated with the undisturbed condition of the ecosystem. Passage, in so-far as it is influenced by human activity, migratory fish;
- the diversity of the higher vertebrate community (amphibians, birds and mammals);
- the structure and quality of the sediment and its ability to sustain the biological community in the ecosystem as well as the riparian and coastal zones, including the biological community and the aesthetics of the site.

The list clearly demonstrates the integrated approach which is chosen to assess ecological quality. It must be stated however that for most target variables standards still have to be developed (nationally). An important shortcoming of the provisional directive is that it does not support the catchment approach which is felt to be necessary (see 2.4).

It should be noted that the directive is in a draft stage and still is under discussion, thus may not come into effect in the referred draft version.

3. Review of biological assessment methods

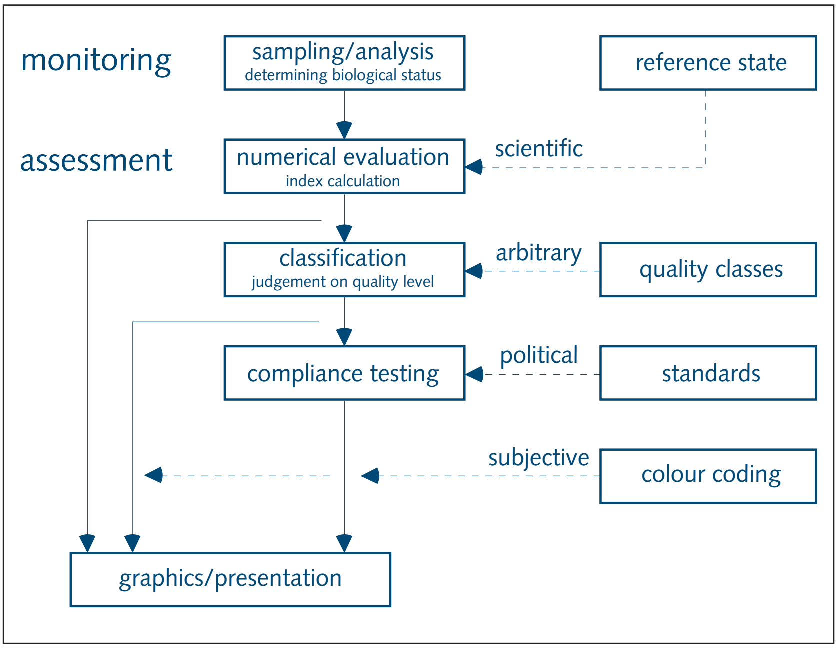
3.1 General

This chapter presents the results of a desk study on biological assessment methods for watercourses. Most attention is given to the state-of-the-art of categories of methods rather than reviewing historical developments of all existing single methods. Description of historical modifications is limited to cases were it is relevant to understanding current practices. As is pointed out in 1.3, biological assessment is handled in its most extended definition, while in geographical respect major emphasis is put on methods which were developed throughout Europe.

The ranking of presentation and the subdivision of the methods in this chapter is a subjective choice which reflects the overall development from the assessment of a single impact on river water (like saprobic systems or toxic impact) to combined or integrated impact assessment of all compartments of riverine ecosystems. Within some of the categories, numerous methods can be distinguished. One or two commonly used methods in routine monitoring however have been presented per category in further detail in the annexes. A fuller description of many methods mentioned can be found in Newman (1988).

Figure 3.1 provides a scheme to describe the process of biological monitoring and assessment in a number of steps. The presented methods are composed of different sets of steps or elements from this scheme. Only a few described methods apply to a complete monitoring and assessment method.

Figure 3.1
Elements of biological assessment methods.



For every group of assessment methods the relevant elements are indicated in dark blue. The elements or arrows, which are not relevant, are coloured in light blue. In some cases an element is optional or is not valid for all methods concerned. This is indicated by dashed elements and arrows.

As can be observed from the illustrations in the following paragraphs, only a limited number of methods comprise all steps in the process of monitoring and assessment. Classification quality levels and presentation methods are available in the saprobic system and biotic indices. Many examples of water quality classes and colour bandings are available on the European continent. Both methods are in principle suited to set standards to be tested for compliance, but only a limited number of examples have been found in literature.

Currently, the European committee for standardization (CEN) is preparing guidelines on presentation of biological water quality data for running waters, using benthic macroinvertebrates. The CEN has noted a great similarity in presentation methods of coloured maps in different countries despite of different assessment methods used. CEN proposes to harmonize the presentation method rather than the assessment method. There is agreement on the following colour coding:

blue	expected natural biological quality
green	slightly impaired biological quality
yellow	moderately impaired quality
red	severely impaired biological quality
black	no macroinvertebrates present, indicating excessive toxicity.

3.2 Considerations on commonly applied biotic groups in biological assessment

Considering the routine monitoring and assessment programmes presented in the literature, it can be concluded that the integrated approach in river quality assessment is a future perspective, whereas the 'classical' biological assessment methods for water bodies are currently used. For this reason, some theoretical and practical considerations on the use of specific groups in biological assessment methods are presented below.

Bacteria

Bacterial methods are applied to assess three different aspects of water quality: hygienic status, mutagenicity and acute toxicity. Microbiological methods in water quality assessment can be considered as a form of biological assessment because of their usage of organisms. In contrast with other biological assessment methods, these methods are however not concerned with the species composition or structure of the bacterial community of the river water, but with the presence of a few indicative species or genera only e.g. pathogenic bacteria. Some other types of bacterial methods involve laboratory tests with well defined strains of a single species, like Photobacterium phosphoreum in the Microtox-test for acute toxicity (De Zwart & Slooff, 1983; Ross & Henebry, 1989).

Algae

Algae have a particular value to assess eutrophication effects, especially in downstream, slowly flowing parts of rivers. Although the existence of a true phytoplankton community has often been debatable in rivers, there is evidence that a dense and true phytoplankton community develops in the middle and lower part of a river provided the residence time is long enough (Tubbing *et al.*, 1994).

In fast running waters and headwaters of rivers, there are hardly any phytoplankton present due to the very short retention time. Attached algae (periphyton) can be used in those cases, but quantitative sampling of this community is very difficult. The use of artificial substrates can overcome this problem. The diatom community can be useful in assessing trophy (Steinberg & Schiefele, 1988). Phytoplankton (suspend algae) are easy to sample in a quantitative way. Identification of species and distinction between living and dead organisms is difficult and can only be performed by trained biologists. Qualitative sampling of periphytic diatoms can be done by scraping off substrates.

Algae exhibit a strong seasonality or periodicity in occurrence, due to the short generation time and variation in competitive power in using the available light. Consequently, sampling frequency should be higher than that for macroinvertebrate community assessment. Algal communities are best suited for assessing the impact of changes in the chemical composition of the water body, rather than physical disturbances.

Macrophytes

Macrophytes are not frequently used in biological assessment of river water quality despite some important advantages: their fixed position and the easy identification. Disadvantages are that they show a strong seasonality in occurrence and visibility. Furthermore, their responses to pollution were not well documented until recently (Hellawell, 1986). In headstreams of rivers macrophytes may be absent, while in lowland streams macrophytes may be often removed by maintenance activities in order to guarantee sufficient discharge.

The above considerations apply to the use in biological assessment of the water body. In adapting a ecosystem approach in the integrated catchment assessment, macrophytes will become of greater importance because of their distribution over all zones of the riverine ecosystem. In the typology of riverine ecotopes an important role has been assigned to plants in characterising the ecotopes (Rademakers & Wolfert, 1994). Macrophytes are important in defining habitat structure and flow for other biotic components.

Macroinvertebrates

The major advantages of using macroinvertebrates in biological assessment have been summarized by Hellawell (1986), Metcalfe (1989), De Pauw & Hawkes (1993):

- the community consists of many representatives from a wide range of faunal orders. It is assumed that such a range of species provides sufficient probability of sensitive species being present;
- spatial and temporal mobility of macroinvertebrates is quite restricted. They can be considered as inhabitants from habitats under investigation;
- organisms integrate environmental conditions over longs periods of time.

Some practical considerations that should be kept in mind when collecting macro-invertebrates concern the seasonality of the presence of a large portion of macroinvertebrate species, namely insects in their larval stage of the life cycle. Furthermore, macroinvertebrates exhibit a large variation in spatial distribution at a specific location.
As a result quantitative sampling is considered to be impossible in routine practice. The use of relative abundances is often applied to get around this problem. Other problems are drift in case of flooding or extreme discharges

and migration or colonisation of exotic species (e.g. in the river Rhine (Van den Brink *et al.*, 1991)).

It has been found that from 100 different existing biological assessment methods, two thirds are based on macroinvertebrates. Three inter-calibration exercises of European methods demonstrated that the most successful assessment methods were those based on the benthic macroinvertebrate community (De Pauw & Hawkes,1993)(see 3.4).

Fish

Fish communities are less frequently used for biological assessment than macroinvertebrates. This is due to some behavioral characteristics of fish. In general fish species are more mobile, e.g. at food collecting, than species of benthic macroinvertebrate community. Apart from this small scale mobility, many fish species show seasonal upstream or downstream migrations for spawning. Fish can show avoidance behaviour to pollution. Another drawback is the necessity of extensive manpower for sampling, especially in deep, fast-flowing rivers (Hellawell, 1986).

Nevertheless, some authors evaluate environmental impact on streams by means of fish community composition and disagree with respect to the sampling effort needed. Karr mentioned some important advantages of using fish communities (Karr, 1981):
- fish are good indicators of long-term effects and broad habitat conditions because they are long-lived and mobile;
- fish communities are composed of several trophic levels (omnivores, herbivores, planktivores
- the position of fish at the top of the predator-prey chain and human consumption make them important key taxa;
- fish provide the possibility of using biomarkers;
- fish are relatively easy to collect and identify to species level (Plafkin *et al.*, 1989).

In general assessment by means of fishes concerns the use of minor fish species rather than commercial fish or ‘angling’ fish.

In European biological water quality assessment some fish species have been implemented in the saprobic system of Sládecek (1973) and can serve as indicators of saprobic load.

Water birds and mammals

As a direct consequence of the classical ‘water body’ approach of biological assessment of watercourses, virtually no attention has been paid in the past to the water birds and mammals as part of a riverine ecosystem. By tradition, water birds and mammals have been the subject of nature conservation institutions rather than water management authorities. For breeding birds, monitoring and assessment is focused on red list or endangered species, whereas for non-breeding water birds the 1% criterion of the Ramsar Convention is applied.

The ecosystem approach for water systems in current Dutch water management, encloses a number of birds as part of the riverine ecosystem, which is visualised by the AMOEBA presentation method. The present abundance of specific water-related bird species is related to the abundance in a historical reference state, specified by a certain year. More attention will be give to this approach in Section 3.8.

Suitability of biotic groups for biological quality assessment

The distinct advantages and disadvantages of biotic groups for monitoring and assessment of river ecosystems which are pointed out above can be briefly summarized into an overall suitability for monitoring purposes for different zones (see table 3.1).

Table 3.1
Suitability of biotic groups for assessment (*separately or in combination*) of distinct riverine zones.
- = not suitable
-/+ = suitability doubtful
+ = suitable
++ = well suitable

	bacteria	algae	macro-invertebrates	macrophytes	fish	birds/mammals
aquatic zone (water body)	++	-/+	++	-/+	++	+
riparian zone (banks)	-	-	+	++	+	++
terrestrial zone (floodplains)	-	-	+	++	-	++

3.3 Diversity indices

Objective

A diversity index aims at evaluating community structure with respect to occurrence of species. Diversity indices relate the number of observed species (richness) to the number of individuals (abundance). Some diversity indices provide an additional insight by calculating the uniformity of the distribution (evenness) of the number of individuals over the counted species. In some cases, diversity is considered to be the species richness only.

Principle

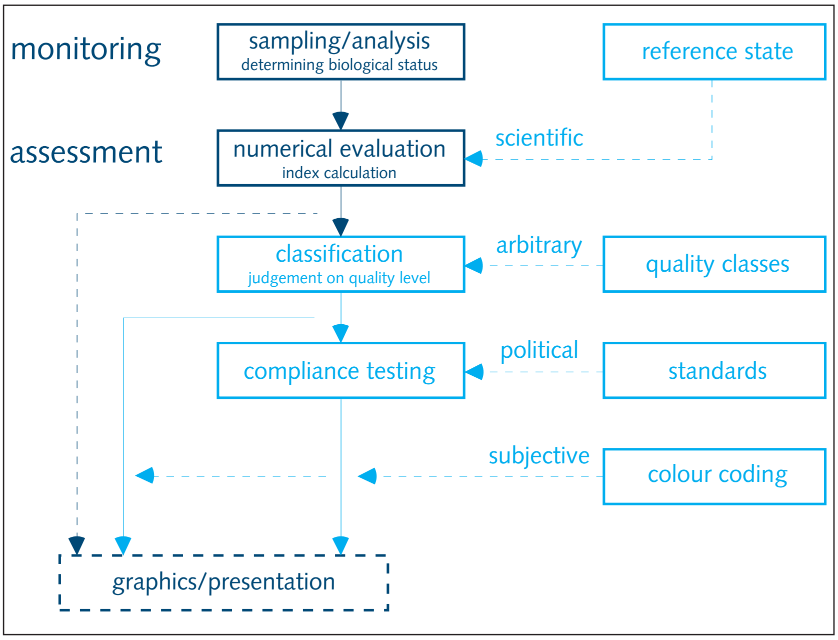
Diversity is a basic feature of the structure of a community or ecosystem, both terrestrial and aquatic (Odum, 1975). The basic assumption is that disturbance of the water ecosystem or communities under stress leads to a reduction in diversity (Hellawell, 1986). Pollution, acting as stressor will result in a reduction of diversity to an extent depending on the degree of pollution. The opposite, low diversity as indication for polluted conditions, is however not necessarily true since low diversity may be caused by other stressors like physical conditions in headstreams (Hawkes, 1979). For similar reasons, temporal changes in diversity at one station are more significant than spatial changes along the longitudinal axis of the river.

Diversity indices can be applied for most biotic groups present in a river. Some diversity indices consider only a part of a community, e.g. ratio of Chironomids and Oligochaetes as part of the macroinvertebrate community (Brinkhurst, 1966). A closely related group of indices that provide information on community structure are comparative and similarity indices. These indices determine to what extent two or more biotic communities resemble each other. They can be used to evaluate spatial discontinuities in communities caused by environmental changes or to detect and measure temporal changes between successive samples.

Scope and limitations

The use of diversity indices in many scientific disciplines may be considered as having world-wide acceptance and application. On a global scale, nature conservation strategies (i.e. Rio Convention) have been formulated in terms of biodiversity (in the sense of species richness). In water quality studies diversity indices often are used in evaluating communities in a ‘before and after’ situation, for example upstream and downstream stations of a wide range of disturbances like discharge of toxic substances (acid mine drainage), nutrient enrichment etcetera.

Figure 3.2
Monitoring and assessment elements of diversity indices.



Diversity indices have some favourable features:
- they are easy to use and calculate;
- they are applicable to all kind of watercourses;
- they have geographical limitations;
- they are best used for comparative purposes.

The principal objections to diversity indices from the point of view of water management and control are:
- they provide information on the biological status without having a clear 'assessment endpoint'. Diversity of communities in natural or undisturbed waters can vary considerably within and in between different water types. The method cannot serve broad surveys over wide ranges of watersheds, due to the great natural variation in physical and chemical conditions (Andersen *et al.*, 1984);
- all species have equal weight, despite known differences in tolerance for pollution, and no information is obtained about the species composition.

Examination of the sensitivity of nine diversity and seven similarity indices shows that the response of the community level indices is dependent on the initial structure of the community, and the manner in which the community is changed (Boyle *et al.*, 1990).
The community level indices may give very misleading biological interpretations of the data they are intending to summarize. Authors state that these indices should never be used alone.

In summary, it can be concluded that diversity and comparative indices are not suitable on their own for routine monitoring of riverine ecosystems at the scale of (transboundary) catchment basins.

Information requirements
Diversity indices can be established by sampling and species identification of a chosen biotic group, mostly macroinvertebrates or algae. The level of identification can vary from species to family level. No specific sampling method or devices are prescribed. It is however essential to use a standard sample and enumeration when comparing impacted sites with a reference site. Sampling strategy concerning density of monitoring station

network and sampling frequency is not dependent on a diversity index as such but is related to the biotic group where it is applied.

Presentation methods

Diversity indices are often presented in a table. Graphical ways of presentation that are suitable for rivers include graphs with the longitudinal distance of the sampling sites at the X-axis and diversity at the Y-axis. The location of impact between stations often is indicated by an arrow. There is no assessment endpoint or reference level that can be referred to.

Examples

Many diversity and comparative indices have been reported (Hellawell, 1986; De Pauw *et al.*, 1992) and evaluated with respect to sensitivity (Boyle *et al.*, 1990). Annex 2 provides a selection. A number of these indices form a part of the metrics in Rapid Bioassessment Protocols that are in use in the United States of America; see Section 3.7.).

3.4 Biotic indices and biotic scores

Objective

Biotic indices and biotic scores are applied to assess biological water quality of running waters, in most cases based on macroinvertebrate community. Biotic indices and scores can measure various types of environmental stress, organic pollution, acid waters etcetera.
The saprobic index can be considered as a specific form of a biotic index. Because of its widespread application, the saprobic index will be covered separately in Section 3.5.

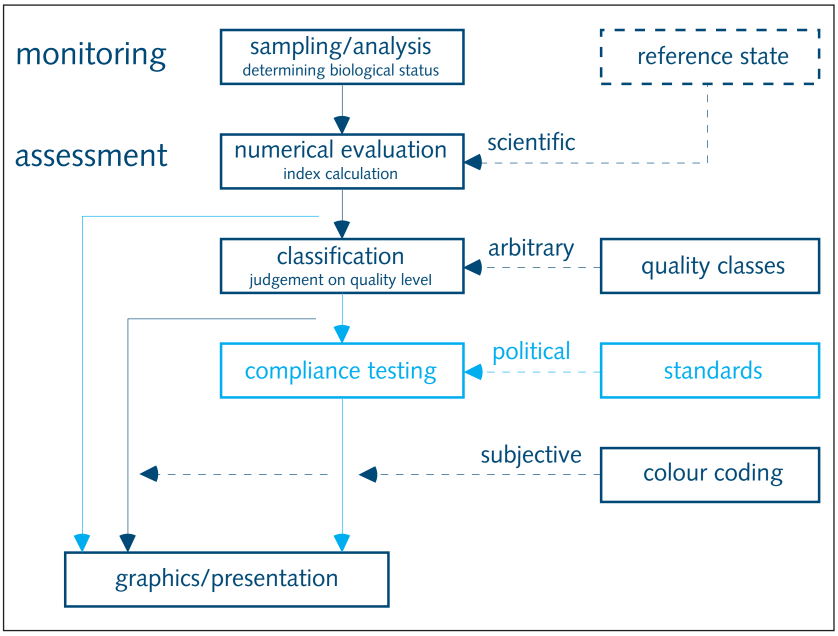
Principle

Biotic score and biotic indices combine features of both the diversity approach (see Section 3.3.) as well as the saprobic approach (see Section 3.5.). The biotic indices are based on two principles: a) that macroinvertebrate groups Plecoptera (stoneflies), Ephemeroptera (mayflies), Trichoptera (caddisflies), Gammarus, Asellus, red Chironomids and Tubificidae disappear in the order mentioned as pollution increases; b) the number of taxonomic groups is reduced as organic pollution increases. A biotic index is a qualitative measure whereas most biotic score includes a measure of abundance and thus is semi-quantitative.

Scope and limitations

The history of the development of biotic indices using macroinvertebrates has extensively been presented and discussed by Metcalfe (1989) (see figure 2.4). Most biotic indices can be considered descendants of the Trent Biotic Index (Woodiwiss, 1964).
Many contributors to the International Conference on River Quality held at Brussels in 1991 presented papers on the use of biotic indices (Newman et al., 1992). Since the late seventies, three rounds of international testing and evaluation of the most common used biotic scores and indices have been performed in West-Germany, United Kingdom and Italy, as an initiative of the EEC (Tittizer, 1976; Woodiwiss, 1978; Ghetti & Bonazzi, 1980). Apart from the wish to develop standard versions of those assessment methods that appeared most practical, some problems remained concerning translation of biotic indices into degrees of pollution, combined with other environmental data like stream velocity, nature of river bottom and climate as well as the need for a biotypological classification of reference biocoenoses.

Figure 3.3
Monitoring and assessment elements of biotic indices.



Most modifications of the original Trent Biotic Index concerned alterations in the groups that determine systematic units. In Denmark however a more principal modification of the Trent Biotic Index was proposed by incorporating two new principles: first, the assignment of negative indicative value to some taxa present and second, the consideration of the number of taxonomic groups as the difference of negative groups and positive groups (Andersen *et al.*, 1984). Thus the utility of the basic principles, increasing pollution results in decreasing number of taxonomic groups, is enhanced. Authors assume that the modified index is applicable to the whole North European lowland.

Some authors state that biotic indices are of an objective type, presenting methods for fixed calculations for any given community, whereas subjective types of (like saprobic) indices depend on the researchers personal interpretation of the fauna in the watercourse present (Andersen *et al.*, 1984). Index values assessed by different persons would be comparable. Hawkes (1979) stated however, that diversity indices are more objective than biotic indices. In biotic indices indicator values are subjectively chosen as in the saprobic system. The biotic index implies more knowledge than actually exists: pollution tolerances are subjective and based on ecological observations and rarely confirmed by experimental studies (Slooff, 1983).

An important advantage of the use of biotic indices is the requirement of qualitative sampling only and identification is mostly at family or genus level, without the need to count abundances per species. Uncertainties in the biotic index only occur due to random variation in samples taken under the same conditions and variation in applied sampling techniques.

A major obstacle in incorporating biotic indices or scores into water management policies and standards is to determine representative reference communities to which investigated stations can be compared. As a result of biogeographical distributions of species and biotypological differences between streams, an optimal biological assessment can only be achieved through regional adaptations (Tolkamp, 1984, 1985). This awareness can be observed in the large number of modifications and variations in biotic scores and indices that have been developed (see figure 3.4). It should be noted however that these adaptations reflect political regions rather than ecological regions. The availability of a European database on reference states based on ecoregions could overcome the problem of relying on regional adaptations.

Information requirements

Virtually all biotic indices and biotic scores are based on benthic macroinvertebrates. Sampling of this biotic group is considered to be possible only in a qualitative or semi-quantitative manner because of the variation in distribution over habitats present. In addition, it is not possible to use one standardised sampling method to cover the full range of upstream headwaters to large and deep rivers in the downstream part of the catchment basin. The applied sampling frequency for biotic indices is directly related to the observed biotic group, the macroinvertebrates. Frequencies range from one to three per year.

Biotic score systems demand more effort and are less practical to use because of the use of abundance, but they may provide more information (Metcalfe, 1989).

Figure 3.4
Biotic indices and biotic scores [References cited from De Pauw *et al.*, 1992]..

Com. = Communities
A = periphyton
D = Diatoms
F = fish
M = macroinvertebrates
P = plankton
V = aquatic vegetation

Biotic indices	Com.	References
Average Score Per Taxon (ASPT)	M	Armitage et al., 1983
Belgian Biotic Index (BBI)	M	De Pauw & Vanhooren, 1983; NBN T92-402
Biol. Index of Pollut. (BIP)	M	Graham, 1965
Biotic Index (IB)	M	Tuffery & Verneaux, 1968
Biotic Index (IB)	M	Tuffery & Davaine, 1970
Biotic Index (BI)	M	Chutter, 1971
Biotic Index (BI)	M	Hawmiller & Scott, 1977
Biotic Index (BI)	M	Winget & Mangun, 1977
Biotic Index (BI)	M	Hilsenhoff, 1982
Biotic Index for Duero Basin	M	Gonzalez del Tanago & Garcia Jalon, 1984
Biotic Index modif. Rio Segre	M	Palau & Palomes, 1985
Biotic Score (BS)	M	Chandler, 1970
Biotic Score modif. La Mancha	M	Gonzalez del Tanago et al., 1979
Biotic Score modif. Rio Jarama	M	Gonzalez del Tanago & Garcia Jalon, 1980
BMWP-Score (BMWP)	M	Chesters, 1980; Armitage et al., 1983
BMWP Spanish modif. (BMWP')	M	Alba-Tercedor & Sanchez-Ortega, 1988
Cemagref Diatom Index (IDC)	PAD	Cemagref, 1984
Chironomid Index (Ch.I.)	M	Bazerque *et al.*, 1989
Ch.I. based on pupal exuviae	M	Wilson & McGill, 1977
Damage Rating	V	Haslam & Wolseley, 1981
Departm. of Environm. Class.	MF	DOE UK, 1970
Diatom Index (IDD)	AD	Descy, 1979
Diatom Index (ILB)	AD	Lange-Bertelot, 1979
Diatom Index (IPS)	AD	Cemagref, 1982-1984
Diatom Index (IFL)	AD	Fabri & Leclerq, 1984-1986
Diatom Index (ILM)	AD	Leclerq & Maquet, 1987
Diatom Index (CEC)	AD	Descy & Coste, 1991
Extended Biotic Index (EBI)	M	Woodiwiss, 1978
EBI Italian modif (EBI)	M	Ghetti, 1986
EBI Spanisch modif (BILL)	M	Prat *et al.*, 1983; 1986
Index of Biotic Integrity (IBI)	F	Karr *et al.*, 1986
Family Biotic Index (FBI)	M	Hilsenhoff, 1987; 1988
Generic Diatom Index (IDG)	AD	Rumeaux & Coste, 1988
Global Biotic Index (IBG)	M	Verneaux *et al.*, 1984; AFNOR T 90-350
Glob. Biot. Qual. Index (IQBG)	M	Verneaux *et al.*, 1976
Ichthygological Index	F	Badino *et al.*, 1991
Lincoln Quality Index (LQI)	M	Extance *et al.*, 1987
Macroindex	M	Perret, 1977
Median Diatomic Index (MI)	AD	Bazerque *et al.*, 1989
Pollution index (I)	M	Beck, 1955
Quality Index (K135, K12345)	M	Tolkamp & Gardeniers, 1977
Quality Rating System (Q-value)	M	Flanagan & Toner, 1972
Simplified Biotec Index (SBI)	MF	Jordana *et al.*, 1989
Trent Biotec Index (TBI)	M	Woodiwiss, 1964

Presentation methods

Calculation of biotic indices and biotic scores result in a number on a certain scale (for example 1-10). In countries that apply an index for nationwide routine monitoring, the value of the index is classified into water quality classes ranging from very poor to very good.
This classification provides the possibility of colour coding of stations or river stretches on geographical maps (e.g. Vlaamse Milieumaatschappij, 1994; Verdievel, 1995).

Examples

De Pauw *et al.* (1992) provide an overview of the biological assessment methods in countries of the European Community. In the majority of cases, these methods are some type of biotic score or index. In almost every country of Western Europe some efforts have been made to test the use of an existing method or a modification of one method or another. This concerns both research purposes and routine monitoring purposes.

As an example of the use of a biotic index in a national routine monitoring and assessment programm, the Belgian Biotic Index will be discussed in detail in annex 3 (Vlaamse Milieumaatschappij, 1994).

In annex 4, the recently developed River Invertebrate Predictions and Classification System (RIVPACS) for the United Kingdom will be discussed. This method uses a concept in which the natural or reference state is predicted for a specific site, deducted from the present value of natural abiotic factors. The macroinvertebrate community which is actually present is compared with the predicted community. Although this method seems to have elements of an ecological assessment method because abiotic factors are involved, it provides no judgement or quality assessment of these factors. Nevertheless, RIVPACS overcomes a disadvantage of biotic scores in general, namely the sensitivity for natural regional differences (Seager *et al.*, 1992).

3.5 Saprobic systems

The saprobic index in the saprobic system could be considered a specific form of a biotic index, but is also often treated as a separate group (Metcalfe, 1989; De Pauw & Hawkes, 1993). Because of some distinct differences and the wide spread application the saprobic index will be covered here separately.

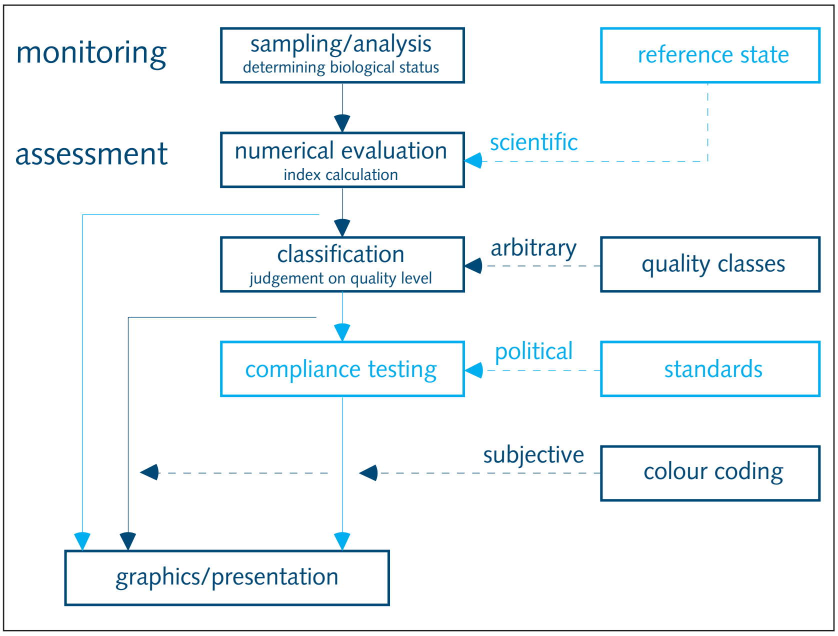
Objective

A saprobic system aims to provide a water quality classification from pure to polluted by means of a system of aquatic organisms indicating by their presence and vital activity the different levels of water quality (Sládecek, 1973).

Principle

The saprobic systems are based upon the observation that species composition as well as species numbers are different over a gradient of self purification after organic inputs, ranging from completed oxidation to predominance of reduction processes. As a result, a zonation in the aquatic communities can be distinguished reflecting the degree of saprobity. Every species has a specific dependency of decomposing organic substances and thus the oxygen content. This (known) tolerance is expressed in a saprobic indicator value, which is assigned to a large number of autotrophic and heterotrophic floral and faunal species.

Figure 3.5
Monitoring and assessment elements of saprobic system.



The saprobity or saprobic index is a numerical evaluation of the presence of indicator species and their respective saprobic values. The saprobic index can be part of a saprobic classification scheme with hydrochemical variables like oxygen content, biochemical oxygen demand or ammonia-nitrogen content, and/or microbiological variables or indices of pollution (e.g. LAWA, 1976; Polishchuk *et al.*, 1984; Aleksandrova *et al.*, 1986; Friedrich,1990).

According to the Pantle & Buck method (1955), each indicator species belongs to a certain degree of saprobity. The saprobic index S can be calculated for a particular subsystem of a biocenose using the following formula:

$$S= \frac{\sum (h_i\ s_i)}{\sum h_i}$$

where
i= number of species, h_i is the quantitative abundance of i-th species (1 = very rare; 9 = mass development) and s_i is saprobic value of i-th species (0 = xenosaprobic, 4 = polysaprobic).

An important objection against this formula is the fact that a species is part of one distinct saprobic zone only, whereas the tolerance usually has a gaussian distribution.

An alternative method is based upon concepts on saprobic valence and indicator weight (Zelinka & Marvan, 1961). To each species a value on a 10-point scale of saprobic valence is assigned. With the use of this method the maximum frequency of the species in a specific zone of pollution is taken into account. The calculation of the saprobity level X is as follows:

$$X= \frac{\sum (s_i\ h_i\ g_i)}{\sum (h_i\ g_i)}$$

where
i = number of species, s_i= saprobic valency of i-th species for saprobity level X, h_i = semi-quantitative abundance, g_i = indicative weight of species (1-5).

Scope and limitations

The origin and historical evolution of saprobic or saprobity indices has been extensively reviewed by Sládecek (1973). The indicator values for saprobity for all species result from empirical data of research in rivers in Central Europe. At present, the saprobic system is mainly used in two ways that differ in calculation method (i.e. the formula of Pantle & Buck or the formula of Zelinka & Marvan) and in applied species indicative values (i.e. the list of Sládecek (1973) or the revised list given in the latest German standard (DIN 38410)).

This revision was based on statistical data analysis of long term biological water quality monitoring. Phototrophic species like algae were excluded because they do not fit into the definition of saproby (heterotrophic intensity). Other criteria for selecting indicator species were: only benthic species are included which reflect the situation of the site; identification at species level should be possible with available keys; the organisms should be spread over Central Europe and finally the saprobic valences should be as narrow as possible (Friedrich,1990). Saprobic systems can differ in the number of distinguished saprobic zones and the index calculation which is used. The system implies more knowledge than actually exists: pollution tolerances are highly subjective and based on ecological observations and rarely confirmed by experimental studies (Slooff, 1983).

Advantages of the saprobic system are:
- quick classification of the investigated community (saprobiological index) can be made on a universal scale from the standpoint of practical use of the water (Sládecek, 1979);
- classification of assessment results are suitable for defining water quality objectives or standards and allow clear presentations in colors on a geographical map;
- the saprobic system can be used in testing for compliance with standards.

Information requirements

The Saprobic index can be obtained for several biotic groups: decomposers (bacteria), primary producers and consumers (zooplankton and zoobenthos/macroinvertebrates). In some countries the Saprobic index S is calculated based on macroinvertebrates (e.g. Germany and Austria) while other countries (also) apply algal species. Saprobic indices are often tied to hydrochemical indices or classifications.

Application of saprobic index requires a qualitative sampling and assessment of abundance of one or more biotic groups. Identification is mandatory at species level because the requirements and tolerances differ for certain species within the same family.

Presentation methods

For the saprobic system several classification schemes are known. Classification of assessment results into a distinct (5-7) number of classes creates the possibility to present results in colours on a geographical map of the river(basin) under study.

Examples

The saprobic system was and is up till now applied in many European countries, e.g. Germany and Austria. In Germany a saprobic system (Saprobiensystem) is in use for routine monitoring and assessment of running waters, as a part of an ecological assessment in water quality maps

(Gewassergutekarte). (In annex 5 this method is briefly introduced, followed by an elaboration on the structure quality assessment).

Koskciuszko & Prajer (1990) applied the saprobic index (formula of Pantle & Buck) in assessing the effect of municipal and industrial pollution on the biological and chemical quality in a Polish river. The Pantle & Buck method in Sládeceks modification has proved to be most convenient for the majority of the investigations (Polishchuk *et al.*, 1984). Authors came to the conclusion that evaluation of water quality based upon phytoperiphyton, phytoplankton, zooplankton and zoobenthos proved to be quite close to each other. In most cases, study of one of these biotic components provided sufficient information for quality monitoring purposes.

3.6 Habitat quality assessment

Although the assessment of habitat quality can not be considered a biological assessment method, attention will be given to this issue in this section because it can be part of ecological assessment.

Objective

Assessment of habitat quality concerns recording and evaluating physical characteristics of watercourses. A specific application assess habitat quality with respect to key species in order to quantify impact on habitats and related species after physical disturbances or rehabilitation measures.

Principle

At present, there are at least two important methods for assessing habitat quality of watercourses, namely:

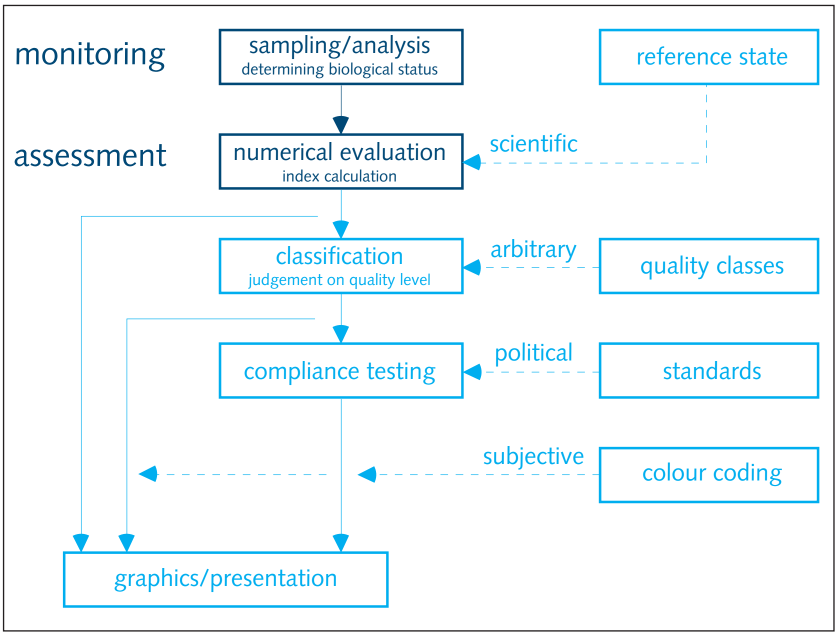
- the Habitat Evaluation Procedure (HEP), developed by the US Fish and Wildlife Service;
- habitat quality assessment as part of an integrated assessment method like in the Rapid Bioassessment Protocols (to be discussed in Section 3.7) or an ecological method like the German stream structure assessment which complements the biological assessment of water quality (to be discussed in annex 5).

The HEP approach is elaborated in this section. Later on in this section (in the examples), attention is also paid to other methods of habitat quality assessment.
The Habitat Evaluation Procedures follow a selective approach for individual key species (US Fish and Wildlife Service, 1980). Abiotic variables of a habitat relevant to the key species are quantified for a specific site under study. This information is related to the known tolerances and preferences of the target species, which are quantified in Habitat Suitability Index models (US Fish and Wildlife Service, 1981). The HSI models calculate the habitat quality, which is a value between 0.0 and 1.0. In a HEP the habitat quality is multiplied by the habitat area, resulting in habitat units, a combination of quality and quantity measures.

Scope and limitations

The Habitat Evaluation Procedures provide a means to quantify the impact of water management measures with respect to loss of habitat structure, and to quantify compensating measures or evaluate rehabilitation measures. Important distinguishing features compared to classical biological assessment methods are the devotion to single key species and the absence of a true assessment classification. The main purpose lies in quantification rather than quality assessment. While at this time HEPs are mainly used in the United States, the application in Europe is currently investigated (TNO, 1992). In future, the method may become a part of an integrated assessment scheme.

Figure 3.6
Monitoring and assessment elements of Habitat Evaluation Procedures.



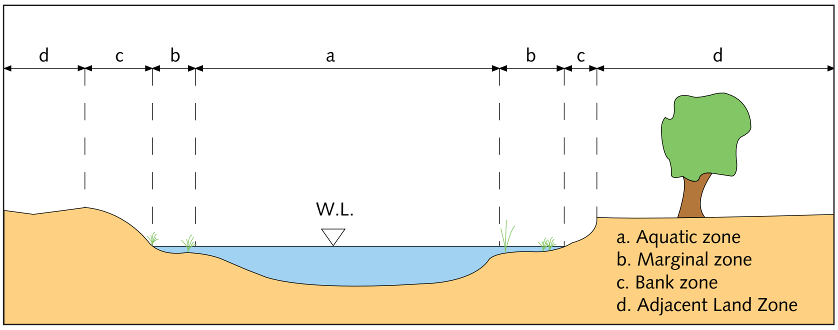
Information requirements

Habitat quality assessment methods all require field inspection and measurements on abiotic variables like stream morphology, substrate types and surface areas, particle size distribution, current velocity etcetera. Furthermore, when applying HEP as many as possible HSI models have to be available concerning the designated key species.

Presentation methods

Results of the field investigations in Habitat Evaluation Procedures can be presented on maps by means of a Geographical Information System (GIS) indicating the suitability of specific areas for the key species. No standard classification could be found in literature.

Figure 3.7
Diagrammatic cross section of a river corridor indicating survey zones [redrawn from National Rivers Authority, 1992].



Examples

On a regional scale, much effort has been devoted to develop methods for assessing the abiotic habitat structure. In Germany and Austria many efforts are in progress to develop water structure maps ('Gewässerstrukturgütekarte') (Friedrich *et al.*,1993) (see annex 5).

In the United Kingdom the River Corridor Survey (RCS) is used as a habitat based tool to evaluate the (potential) impact of land drainage and flood prevention measures on bird, mammals and riparian invertebrates. The background is being formed by wildlife and nature conservation guidelines (Rheinallt, 1990). Only in limited regions is the habitat information related to species information. The technical methodology of RCS includes the recording of major habitats in four zones of the riverine ecosystem: the aquatic zone, the marginal zone, the bank zone and the adjacent land zone (see figure 3.7; redrawn from National Rivers Authority, 1992).

3.7 Rapid Bioassessment Protocols (RBP)

A fairly recent development in biological assessment in the United States is the use of Rapid Bioassessment Protocols. The approach was first developed by Karr (1981) for fish communities and later refined. The US Environmental Protection Agency developed in 1989 the so-called Rapid Bioassessment Protocols

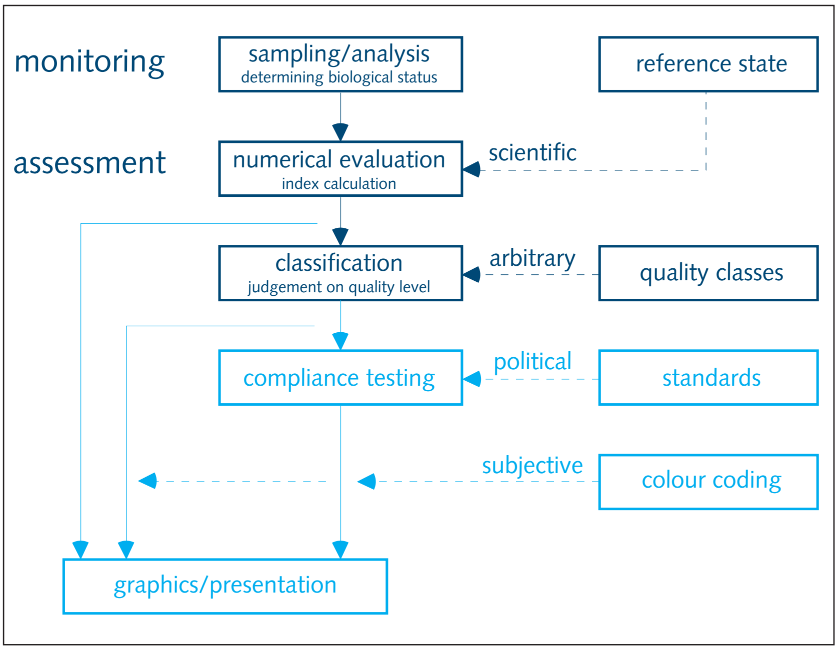
Objective

The objective of RBP is the assessment of ecological integrity and impairment of streams, using macroinvertebrate and/or fish communities.

Principle

Rapid Bioassessment Protocols combine the assessment of the biological condition or quality with the assessment of habitat quality (see figure 3.7). This combined evaluation implies that the method can be considered as an ecological assessment method (Plafkin *et al.*, 1989). Five protocols have been designed, increasing in complexity and sampling requirements and thus improving assessment results, depending on the desired purpose.

Figure 3.8
Monitoring and assessment elements of Rapid Bioassessment Protocols.



RBP IV consists of a certain number of diversity indices and a number of comparative indices, called metrics (see Section 3.2.). These metrics assess the biological condition of the benthic community, divided in three categories: structure, community balance and functional feeding group

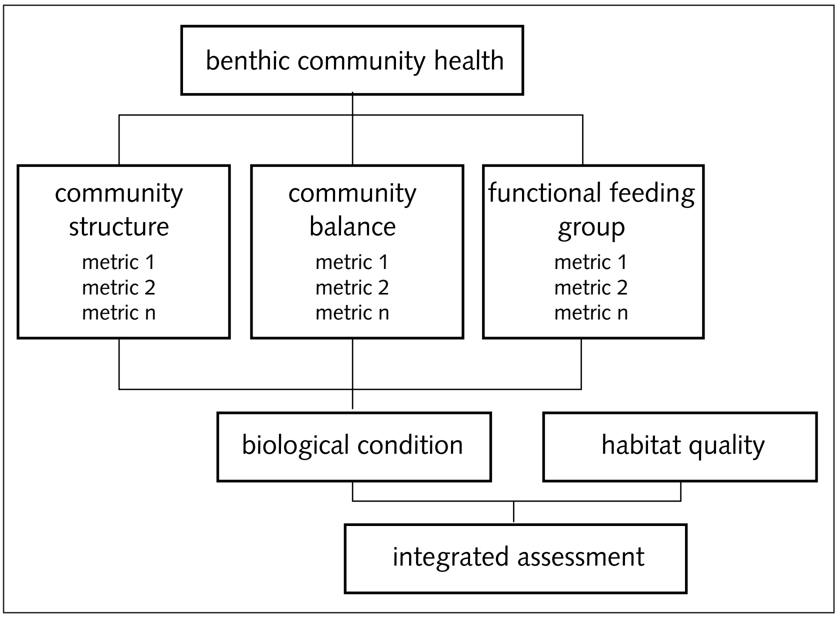
(Barbour *et al.*, 1992). Habitat quality concerns factors like substrate and instream cover, channel morphology and riparian and bank structure.
The major governing principle of the method is the comparison with a reference site or a set of reference data. This principle is comparable with RIVPACS. The difference lies in the nature of the reference site: in RIVPACS this is site-specifically predicted, in RBP this is a mean situation of a set of unaffected sites. It is concerned with the habitat structure available to macroinvertebrate or fish community, compared to the habitat structure of a reference site under natural conditions. This results in a percentage resemblance, which can be classified from poor to excellent. Afterwards the habitat quality is evaluated in combination with the biological condition of the communities, resulting in an integrated assessment (see Section 3.6).

Scope and limitations

Application of Rapid Bioassessment Protocols is found to be limited to the United States. In a number of States (e.g. Ohio, Arkansas, Illinois, New York) the basic approach of this method has been tested and further developed. No examples of application in Europe have been found in literature.

Barbour *et al.* (1992) published an extensive evaluation of the metrics concerning biological quality with respect to redundancy and variability among metrics for a set of data from reference streams. They proposed to restrict the number of metrics because of occurring redundancy and dependency.

Figuur 3.9
Conceptual base for Rapid Bioassessment Protocols [after Barbour *et al.*, 1992].



Information requirements

Most biological metrics use the benthic macroinvertebrate community and in some cases the fish community. Calculation of the metrics requires standard sampling techniques, which are extensively described and accompanied with guidance and data sheets.
The identification of macroinvertebrates is required at the family level, while abundances can be estimated in a qualitative manner. As a result the assessment can be considered ‘rapid’.
It should however be noted that the RBP's are not more rapid that biotic indices.

The variables concerning the habitat quality are also compared with a reference site and results are give as a percentage resemblance, which in turn is classified in one of four classes from non- impaired to severely impaired.

Presentation methods

The results of RBP is presented as numbers in a table (Plafkin *et al.*, 1989).
A guidance on (graphical) presentation methods (e.g. by means of coloured classification) is not given.

Examples

In literature no examples of application outside the United States of America have been found.

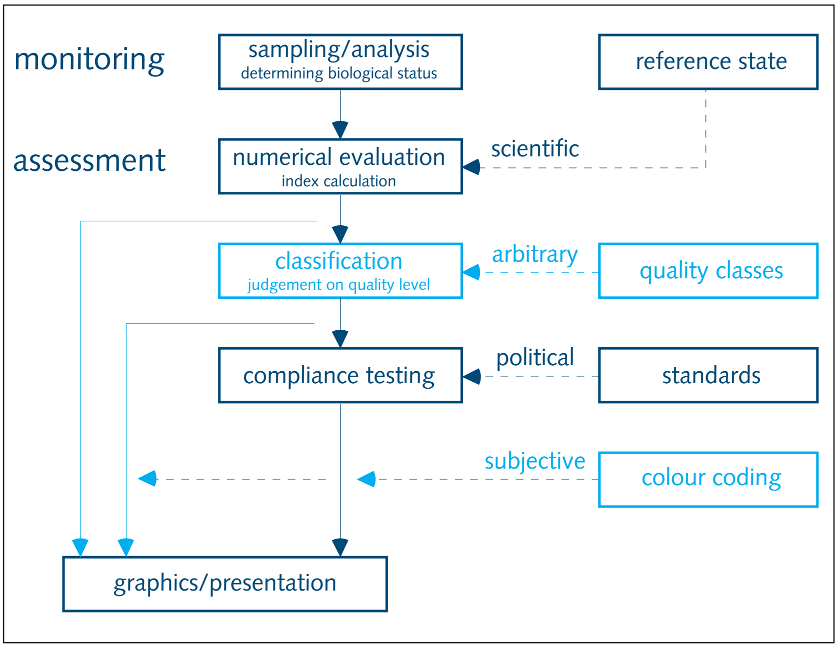
3.8 Ecosystem approach in integrated water management

Ecological water quality assessment shows three important characteristics (Klapwijk *et al.*, 1994):

- an approach which includes the functioning of the whole aquatic ecosystem considering a number of abiotic and biotic components and their interrelations;
- a multilateral approach instead of an approach from only a limited number of influencing factors such as saprobity and trophism. As many as possible factors, affecting the characteristics of water types, are involved;
- ecological assessment provides the water manager with special tools in order to steer an aquatic ecosystem in the desired direction.

Based on these characteristics, an ecological assessment method for Dutch running waters has been developed, based on macroinvertebrate community (STOWA, 1992).
Further details about this method are given in Annex 6. At the moment, the method can be applied to all regional, small waters in the Netherlands. For large rivers, the AMOEBA approach is followed.

Figuur 3.10
Monitoring and assessment elements of AMOEBE approach.



Principle

In the ecosystem approach the river is considered to be a part of a drainage basin water system which includes both water body, bottom, banks and the terrestrial zone (Laane & Lindgaard-Joergensen, 1992; Schulte-Wülwer-Leidig, 1992). As a result of the complex interactions between both abiotic determining factors and biotic components, an ecosystem approach is needed in monitoring and assessment to provide information on water management objectives and human uses. The goal of integrated management is to maintain healthy ecosystems in which sustained use by man is possible. Minimal human influence is expected to provide a natural or reference situation, which can be pursued.

Policy objectives for water systems have to be verifiable. Sufficient information is needed on reference state, as a management target, as well as the actual situation. Deviations of the actual status with respect to reference or objective, should be measurable.

For Dutch water management, the AMOEBA (acronym for General Method of Ecosystem Description and Assessment) has been developed (Ten Brink *et al.*, 1991) to evaluate the measured states. The AMOEBA-approach is based on the assumption that an ecosystem which is not or hardly not manipulated, offers the best guarantee for ecological sustainability: the reference system. The introduction of a reference provides a standard by means of which an assessment of the ecological condition of a system can be made.

Information requirements

Information on the reference situation has to be available, whether from historical sources or the river system itself or from analogous situations in other places. To provide information on the present state, monitoring is required, concerning chemical, physical and biological parameters. The parameters should be representative for the ecosystem or ecosystem compartment. The reference values for the arms of the AMOEBA are not necessarily representing the same year.

Presentation methods

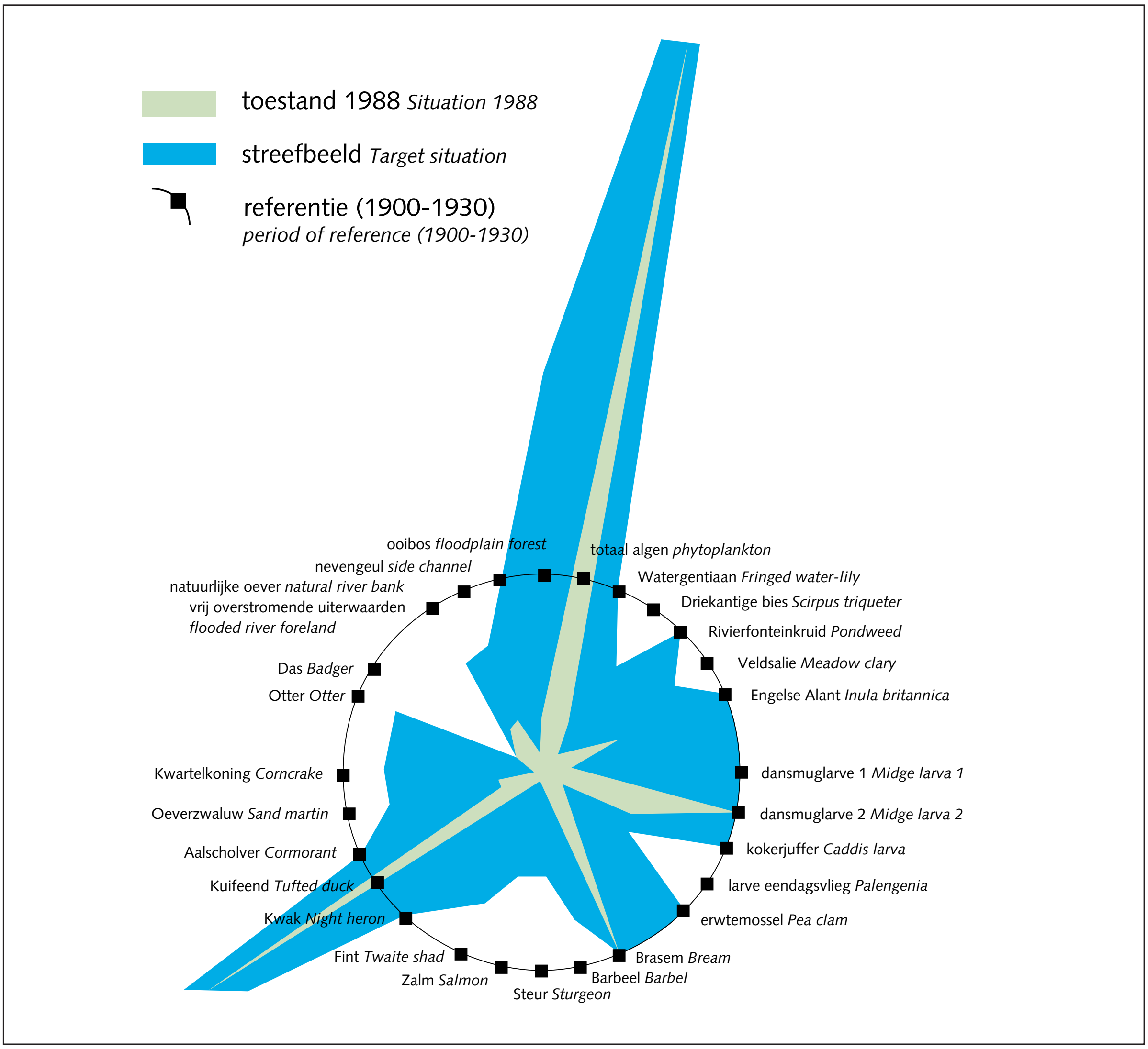
The AMOEBA provides a special method to present or visualize in a graphical way the quantitative relation between reference situation, target situation and present state.
The AMOEBA model is thought to be of practical use to policy makers and decision makers, as a large amount of gathered data is comprehensively summarized and visualised.

Figure 3.11 shows an example of a river AMOEBA (reprinted from Van Dijk & Marteijn, 1993). The targets are in blue, the present state is light coloured whereas the reference situation for each target variable is at the circle. The target value is a political choice and need not necessarily be equal to the reference. It can be somewhere between the present situation and the reference.

The described ecosystem approach and presentation method do not reveal the underlying methodology to measure biological and ecological variables. The AMOEBA focuses on a number of key species for which abundance may be the assessed variable. On the other hand, the method provides a strong facility to apply totally different methods, e.g. for different variables from ecological or ecotope models or Habitat Evaluation Procedures.
An important distinctive feature compared with other biological assessment

methods is the ability to present assessments of different zones of the riverine ecosystem together.
This could be a feature of major importance in the perspective of integrated catchment assessment as an integrating presentation tool. This approach is being applied in the Netherlands for the lower part of the Rhine (Van Dijk & Marteijn, 1993). It should be noted that an AMOEBA has to be accompanied by a specification of the part of the riverine ecosystem (location, stretch, whole river) that is concerned.

Figure 3.11
Example of a river AMOEBA [from Van Dijk & Marteijn, 1993].



3.9 Methods concerning ecosystem functioning

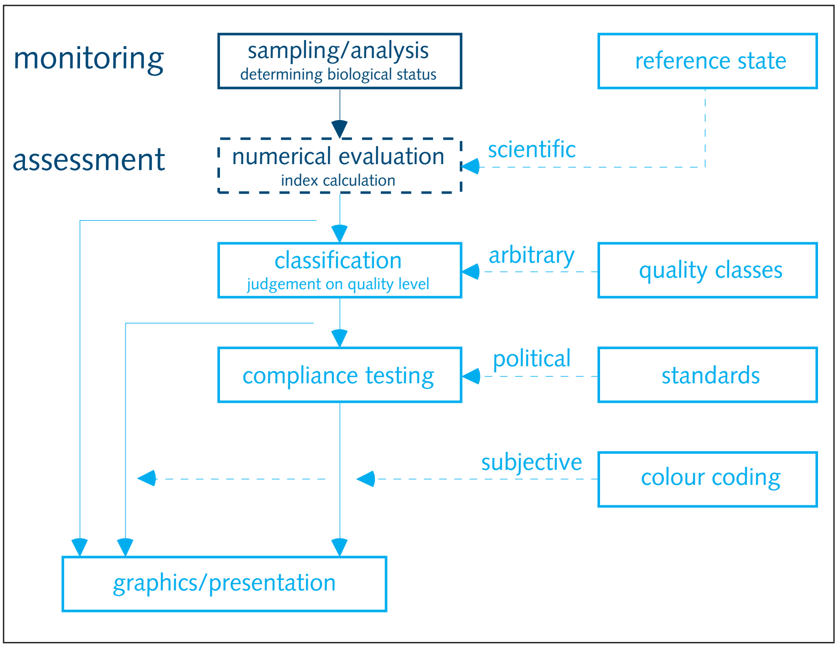
Principle

All assessment methods mentioned above concern the structure of communities of the aquatic ecosystem. Another essential feature is the functioning of the ecosystem.
This regards the processes that take place in the ecosystem, like primary production or gross, primary and secondary consumption, mineralisation or degradation.

Functional features can be studied on the scale of responses of individuals (e.g. Scope for growth test (Bayne *et al.*, 1985)) and of whole communities (e.g. P/R ratios (Odum, 1975), the autotrophy index of Weber (Matthews et al., 1980) and total community respiration (Grimm & Fisher, 1984; Maltby & Calow, 1989). Some aspects of ecosystem functioning can be pointed out as implicitly underlying structural methods like the saprobic system.

There might be discussion as to whether some standard analyses variables in routine monitoring and assessment are to be considered functional or structural variables, for example biomass and chlorophyll-content. In this report, these are regarded as functional, because the variables are considered a measure for the intensity of primary production.
The variables offer however only fragmented information on the ecosystem functioning and are not often implemented in a coherent model of ecosystem functioning.

Figure 3.12
Monitoring and assessment elements of functional methods.



Another example of a functional method based upon an experimental set-up, is the Algal Growth Potential test (AGP). In this bioassay a test organism e.g. the green algae Scenedesmus quadricauda, is used to determine the availability of nutrients for algal growth and to determine the growth limiting nutrients (Klapwijk *et al.*, 1988). In the international standard Selenastrum capricornutum (=*Raphidocelis (Pseudokirchneria) subcapitata)* is applied (ISO 8662).

Examples

In a comparative study of 8 river longitudinal stretches of Lower Dnepr in Russia, an evaluation of water quality was made on the basis of benthic invertebrates using Pantle & Buck's Saprobic index and some functional indices like Gross primary production (P), Destruction of organic matter (D) and P/D ratio (Aleksandrova *et al.*, 1986). Authors found the results of both methods to be closely coinciding. However different indices did not always result in the same classification for an individual stretch.
Aleksandrova *et al.* feel that a combined evaluation gives a unique answer

about the water quality of the stretch while deviations in indices make it possible to judge characteristics of the pollution.

A comparison between several pollution assessment methods in three Belorussian rivers showed that functional phytoplankton indicators (like chlorophyll-a content and phytoplankton production rate) reflect the pollution of the water and its level of self purification much better than bioindicators of the Zelinka & Marvan and Pantle & Buck system (Mikheyeva & Ganchenkova, 1980).

Bombówna & Bucka (1972) applied a bioassay technique to characterize the potential productivity of Carpathian rivers, in order to forecast eutrophication effects after dam construction in the reservoir. They showed with this method that rivers with a similar chemical composition may have different influence on the growth of algae and that bioassay techniques provide additional information to chemical analyses.

Detchev proposes a theoretical model (called functional-ecological approach) based on functional characteristics of the aquatic ecosystem (Dechev *et al.*, 1977; Detchev, 1992).
A major problem with evaluation of biological effects by assessing biological responses is the strong non-linear dose-effect dependence. For this reason, variables such as species composition, diversity and community structure are not sufficient for describing river water quality. Structural variables can not provide information about rates of processes because there is not a linear or even constant dependence between biomass and metabolic rates in ecosystems.

The functional-ecological approach operates with direct in situ measurements of rates of basic processes forming the turnover of substances and energy in the ecosystem and the balance concentrations of important substances. The theory is based on the properties of an optimal self-regulating system (biochemical reactor). The proposed functional approach still is under development and seems not to be available for routine purposes so far.

3.10 Assessment of toxicity, bioaccumulation and mutagenicity

Often disturbance of biological water quality in biological assessment is expressed in terms of ‘pollution’, without specifying the substances involved. Biological assessment methods, like biotic indices (see Section 3.4), do not offer the possibility of discriminating between effects of organic and toxic compounds. The indicative value of species in terms of pollution tolerance was deducted from correlative field observations, supported by chemical analysis, limited to saprobic and trophic compounds. Slooff (1983) indicates that those values have not been confirmed or validated by experimental studies.

Currently available methods in assessing river water or sediment toxicity can be categorized as follows:
- in stream or in situ observations on communities, comparable with other biological assessment systems, to identify effect of toxic substances (3.10.1);
- in stream bioassays (active monitoring) (3.10.2);
- laboratory toxicity tests (bioassay) to assess acute and chronic toxicity to single species (3.9.3);

- bioaccumulation monitoring (3.10.4);
- integrated toxicity assessment (3.10.5).

Mutagenicity is considered to be a specific toxicological effect of substances and is briefly discussed in Section 3.10.6.

3.10.1 In stream observations on communities

At this moment, there are very few available biological assessment methods explicitly based on effect evaluation of toxicity to field communities. This is due to the fact that it is often not possible to discriminate between the effects of toxic substances on organisms and populations and other abiotic factors, that govern the presence of a community.

In most cases, experimental settings are used (see Section 3.9.3.) to assess toxicity, and only in case of sediment quality evaluation are field evaluations being used (see Section 3.9.5.). Investigations on the impact of toxic load on benthic communities often apply diversity indices in comparing upstream and downstream locations of a discharge. It can be difficult to discriminate between natural changes in community compositions along the longitudinal axis of the river and the anthropogenic changes.

Index of Community Structure (ICS)

Clements *et al.* (1992), developed the Index of Community Structure (ICS) to assess toxic impact. This method is based upon experimentally assessed sensitivity data for specific species and toxic substances (heavy metals). The data were obtained from outdoor stream mesocosms. Sensitivity of a species was determined, after colonisation on artificial substratum in the field, by measuring the reduction in abundance (relative to control stream) at a given concentration in the mesocosm. The ICS is given as:

$$ICS=\sum s_i x p_i$$

where sensitivity s_i is proportion reduction in abundance of i-th species in treated streams relative to controls and p_i is proportion abundance of i-th species in field samples.

The method is under development and has not yet been applied for routine monitoring purposes. The single-purpose character of the method, like e.g. saprobic index, seems to prevent the method from becoming an alternative for routine monitoring and assessment over a broad range of streams and impacts. Two important limitations are:
- sensitivity estimates are obtained from a single set of experiments during a short period of exposure and a specific community;
- sensitivity estimates are obtained for a single compound. The large number of toxic substances make it impossible to develop this method for broad application.
  The authors have already simplified the model by assuming that toxicity of metals is fully additive and thus can be totalled.

3.10.2 In stream bioassays

This method concerns the placement of living organisms from a laboratory culture or an uncontaminated water into a river. The advantage of this method is that it provides a direct way of measuring the response of aquatic organisms to present water quality and possible presence of toxic substances. Examples can be found in Seager *et al.* (1992).
They placed the freshwater amphipod (*Gammarus pulex*) in cages upstream and downstream different of types of effluent discharges.

Lethality in case of severe pollution or sublethal effects in case of mild pollution can be studied. A sublethal test is the ‘Scope for Growth’ test. This technique is based on the assumption that organisms stressed by the presence of pollutants have less energy available for growth and reproduction because either a reduction in feeding rate efficiency or a diversion of available resources into preventing or repairing damage. The food consumption and respiration of the caged animals is measured.

3.10.3 Laboratory toxicity testing

In the field of ecotoxicology, numerous single species laboratory tests have been designed to assess toxicity of aqueous solutions like river water or sediments. In principle, test organisms (mainly from standardised cultures) are being exposed to river water or sediment that has been transferred to the laboratory. A very wide array of organisms from all trophic levels are in use ranging from bacteria and algae to fish. The most widely used are species of the waterflea Daphnia. Depending on the test organism, many effect parameters can be distinguished, e.g. lethality, reproduction, development etcetera. In literature many overviews of toxicity tests can be found e.g. Buikema *et al.*, 1982, Boudou & Ribeyre (1989), Hill *et al.* (1993), Phillips & Rainbow (1993), De Zwart (1994). Also some ISO standards are available (ISO 6341:1988 ; 7346-1/2/3: 1984).

A principle drawback of toxicity tests is the problem of extrapolation of the observed effects under laboratory conditions to effects that can be expected under actual conditions in the field or stream.

3.10.4 Bioaccumulation monitoring

The objective of monitoring bioaccumulation is to determine the actual bioavailability of (a mixture of) toxic substances governed by the conditions at a give site. Accumulated concentrations of substances in whole animals or specific tissues are related to environmental concentrations. In determining bioaccumulation a biological and a chemical aspect can be distinguished. Because of the use of living organisms the assessment can be considered a form of biomonitoring. The subsequent analysis methodology of substances in the organism can be considered as a chemical method and will not be discussed here.
Four types of bioaccumulation monitoring can be pointed out : active bioaccumulation monitoring (in stream), passive bioaccumlation monitoring (in stream), laboratory setup and simulating methods.

active biomonitoring

Active biomonitoring is performed by collecting animals from unpolluted locations and afterwards exposing them (in cages) in field situation at a polluted station during a certain period of time. A typical advantage is the possibility of standardising methodology with respect to exposure duration, selection of collected animals by age, size or uniformity.
Often used animals are freshwater mussels (for example Dreissena polymorpha, Anodonta anatina) because their ability to resist high levels of toxic substances and their wide distribution of occurrence (Hemelraad *et al.*, 1986).

A major disadvantage is the seasonality in the availability of suitable organisms at an unpolluted collecting station. When using mussels during the spawning period, the biomass of the mussels is not constant. Furthermore one has to be certain of the absence of pollution at the collecting station. It should be kept in mind that the actual occurrence of the collected species at the polluted site is not a necessity.

passive biomonitoring

Passive biomonitoring is performed by collecting organisms from a particular station at which they are exposed to toxic substances at present and in the past. Disadvantages are clearly the lack of knowledge about the duration of exposure and presence of the animals and the variability in age, size and available number of individuals of the sampled population.
For the performance of the analysis, a minimum amount of tissue has to be available. Also, the toxicity in the water system may be at a level in which the preferred species cannot survive. Numerous examples in literature can be found, e.g. Timmermans *et al.* (1989). In the Dutch routine monitoring program for inland waters, Eel (*Anguila anguila*) is used to assess bioaccumulation.

laboratory experiments

Assessment of bioaccumulation in the laboratory is performed by means of bioassays with field-contaminated water or sediments. Test organisms from a standardised culture are exposed to this water or sediment under controlled conditions (examples: Hill *et al.*, 1993; Hemelraad, 1988).

simulating bioaccumulation

From more recent date a method is under development in which bioconcentration of complex mixtures of hydrophobic substances in river water is simulated. The bioconcentration is simulated by equilibrium partitioning of these compounds onto 'empore disk' a filter material containing a solid phase. Water samples are stirred with a piece of this disk for 14 days. Afterwards the disk is eluted and the extract is concentrated, followed by chemical analysis of total compounds (Verhaar *et al.*, 1994).

3.10.5 Integrated toxicity assessment

To overcome the drawbacks of both field observations and laboratory tests, an integrated approach is proposed in which both aspects are evaluated in combination with chemical analyses. In the Netherlands, the Sediment Quality Triad (SQT) was introduced for integrated assessment of polluted sediments, following the original development in the United States (Van de Guchte, 1992). The SQT considers three components: bioassays in laboratory, field observations on communities and chemical analysis.

In the past few years, the Dutch Ministry of Transport and Waterworks has applied this method to a large number of suspect sites, especially in the sedimentary zone in the downstream regions of the main rivers. Hendriks (1994) presents an integrated assessment for river water quality which is based on the same principle.

3.10.6 Mutagenicity

Mutagenicity testing of river water can be performed by laboratory analysis of river water samples. 'In stream' methods exist of determining incidence of diseases or morphological deviations of organisms in a community, for example tumor incidence in bottom dwelling fish.

Mutagenicity tests can be performed with the aid of bacteria (Ames-test, Mutatox (Ho & Quinn, 1993); De Zwart, 1994) or fish (Sister Chromatid Exchange test with Notobranchius rachovii; (Hendriks, 1994)). A fairly recent mutagenicity test is the UMU-test, which resembles the Ames test and is currently under discussion in European standardisation committees (CEN).

3.11 Microbiological assessment of hygienic status

Objective

Microbiological methods aim at determining the presence of pathogenic bacteria to assess the hygienic status and potential risk to human (and animal) health.

Principle

Surface water can carry a number of different pathogenic organisms due to discharge of (treated) domestic and agricultural waste water. Monitoring the hygienic status of surface water is performed with microbiological water tests. In general these tests involve enumeration of the virulent organisms and identification of special organisms indicative of hygienically suspect contamination or even pathogens themselves. Of the pathogens and facultative pathogenic types which can occur in water, the bacteria of the family Enterobacteriaceae are of special importance. The species Salmonella, Shigella, Escherichia, Erwinia as well as the so-called coliform bacteria belong to this family. Salmonella and Shigella are classed as being particularly pathogenic, while the others being classed as facultatively pathogenic.

Scope and application

Most monitoring programmes of river water quality contain microbiological analyses in order to obtain information on the hygienic state of the water. In general, the assessment is performed in relation to functional uses like recreation, agriculture or drinking water supply. Faecal streptococci (Enterococci) are considered to be the best indicators for human and animal faecal contamination. They rarely multiply in water.

The sampling and enumeration methodology can be applied in all types of running waters throughout Europe. There are no limits in the application because of biogeographical distribution of species.

Information requirements

Sampling methodology for microbiological purposes has been internationally standardised (ISO, 1990). Frequently used species or genera in microbiological assessment are Escherichia coli (E.coli), Salmonella and faecal Streptococci. Differences in methodology for selection of species or species groups involve the temperature of incubation (22, 37 or 44 °C) and incubation media.

Presentation method

Results of microbiological methods are mostly expressed as number per unit of volume and presented in tables.

3.12 Summarizing overview

Table 3.2 provides a tentative summary in which some characteristics of the assessment methods in general and some single methods in particular have been brought together. Information on sampling strategy like methodology and frequency as well as the taxonomic level of identification is strongly dependent on the community observed. To avoid complexity, these have been left out.

Table 3.2
Tentative summary on assessment methods.
Legend:
zones: AQ= aquatic; sd= bottom sediment;
RP = riparian/amphibious zone,banks;
TE= terrestrial zone (floodplains)

communities: A = algae, B= bacteria; F= fish,
M= macroinvertebrates, BI= birds,
MM= mammals, V= macrophytes

* = provided regional differentiation

group	general method and examples	assessment purpose	riverine zone	ecosystem - structure (S) - functioning (F)	level of application: regional (R) national (N)	stream order	community	suitability for use in an integrated catchment approach
diversity and comparative indices		impact on community structure (aspecific)	AQ (,RP,TE)	S	RN	1-10	AMF	-
biotic indices/scores		biological quality	AQ	S	RN	1-8	MD	+ *
	BBI (annex 3)	biological quality	AQ	S	N	1-6?	M	
	RIVPACS (annex 4)	biological quality	AQ	S(F)	N	1-8?	M	
saprobic index		degree of saprobity	AQ	S(F)	N	3-10	BAMF	+
toxicity		toxicity/ bioaccumulation	AQ (+sd)	F	RN		ABMF	+
habitat quality	RCS (river corridor survey); German stream structure assessment.	habitat structure	AQ/RP	S	RN	?	-	-
	HEP	suitability of habitat for key species	AQ/RP/TE	S	R	?	M,F,BI,MM,V	
Rapid Bioassessment		ecological integrity	AQ	S	N	1-?	M,F	+
Protocols								
Ecosystem approach	AMOEBA	integrated management	AQ/RP/TE	S	N	high orders	AFMV, BI/MM	+
	STOWA	ecological quality	AQ	S	N	1-5	M	-
Functional methods		ecosystem processes	AQ	F	RN			+

stream order:

river size	average discharge (m³/s)(km²)(m)	drainage area	river width	stream order **
very large rivers	> 10,000	> 10^6	> 1,500	> 10
large rivers	1,000 - 10,000	100,000 - 10^6	800 - 1,500	7 to 11
rivers	100 - 1,000	10,000 - 100,000	200 - 800	6 to 9
small rivers	10 - 100	1,000 - 10,000	40 - 200	4 to 7
streams	1 - 10	100 - 1,000	8 - 40	3 to 6
small streams	0.1 - 1.0	10 - 100	1 - 8	2 to 5
brooks	< 0.1	< 10	< 10	1 to 3

** depending on local conditions
(from Chapman, 1992)

4. Current practices

4.1 General

This chapter provides an overview of the results of the enquiry among UN/ECE-Task Force-countries with respect to the routine biological surface water monitoring and assessment of rivers. Details on other elements of the current practice on monitoring and assessment can be obtained from part II of these series of reports.

It should be kept in mind that monitoring programmes differ in different rivers within one country with respect to objectives, variables, sampling frequencies etcetera. Furthermore, the enquiry was restricted to transboundary rivers and provides limited or no information on national monitoring programmes in other rivers or watercourses. It has to be noted that the level of detail in the questionnaires shows very large differences between countries making balanced comparisons very difficult.

In Section 4.2, a general description of current practices is presented for the ECE-countries which responded to the questionnaire. The following sections provide summarizing tables on different elements of biological monitoring and assessment. Additionally, comparisons are made between current practices and the state-of-the-art of biological assessment as described in chapter 2.

4.2 Biological assessment practices in ECE-countries

AUSTRIA

Criteria for the routine biological surface water monitoring programmes are set by international commissions for transboundary rivers. The objectives of the programmes are to classify water quality, to collect information with regard to implications of waste water impacts and to perform saprobiological investigations. The variables are most extensive on the Danube, including four to twelve samples a year for microbiological variables and biological structure of phytoplankton and phytobenthos, zooplankton and invertebrate fauna. For the other reported watercourses, the set of variables is less extensive and the frequencies are lower. A 'biocoenotic analysis' is performed on the biotic groups, resulting in a saprobic index. Various species indicator lists regionally used in Austria have been revised and summarised into a catalogue Fauna Aquatica Austriaca (Moog, 1995). This catalogue includes indices of saprobity, functional feeding group classification and expected zonal distributions following biocoenotic regions for Ciliates and selected macrozoobenthic groups; all specifications are at species level. A new guideline for the ecological survey and evaluation of running waters (Onorm M 6232-1995) has been completed recently.

BULGARIA

Routine biological surface water monitoring is not yet applied in transboundary rivers, but a biological assessment method by means of macroinvertebrates is under development. The method will be based on hydrobiological, toxicological and microbiological analyses and aims at assessing impact of pollution on water biota. In some Bulgarian rivers other than transboundary rivers, biological assessment is performed by application of a biotic index and/or a saprobity index.

CZECH REPUBLIC

Biological analysis of bioseston, followed by saprobic evaluation is currently used in the routine surface water monitoring network. In addition, regular monitoring by means of macrozoobenthos analysis is applied in running waters (a special network of several hundreds sites) with a sampling frequency of once per 5 years.

Chlorophyll-a analyses are performed occasionally, however, the regular monitoring in the national network is to be introduced from 1995. Toxicity tests (mainly Daphnia, fish and algae) are performed on selected localities, usually as a special investigation as well as Ames' mutagenicity tests. Detailed biocoenotic analyses are performed in the framework of the river-basin projects, i.e. the Elbe river, Oder river and Morava river with a frequency of once per 3 years. These investigations also include bioaccumulation of pollutants in biomass (fish, macrophytes, macroinvertebrates).

Microbiological analyses are based on determination of coliform bacteria and faecal (thermotolerant) coliform bacteria in standard national network for surface waters. Occasionally heterotrophic bacteria (meso- and psychorophilic plate counts) and faecal streptococci are also determined. Data are produced mainly by laboratories of the River Boards Ltd., while national database of the parameters is held in the Czech Hydrometeorlogical Institute (Prague).

CROATIA

For routine biological monitoring and assessment in Croatia a saprobic system is applied. The saprobity index S according to Pantle & Buck is calculated for phytoplankton, zooplankton and invertebrate fauna using the species indicator value list of Sládecek (1973). Sampling of all three biotic groups is performed by filtration of 50 litres over a plankton net. Apart from saprobiological classification of water quality, cluster analysis is performed to investigate biological structure with emphasis on Eubacteria, Diatomaeae, Chlorophyceae, Cyanophyceae, Rotatoria, Nematoda, Amphipoda, Cladocera, Cnidaria, Oligocheata, Copepoda, larvae and Protozoa.

ESTONIA

Estonia reports routine biological surface water monitoring only for lakes. There is no reporting on this subject for transboundary rivers.

FINLAND

Routine biological surface water monitoring is limited to microbiological analyses. Thresholds for these variables are implemented in water quality criteria for recreational use, fishing water, raw water supply and general water quality. This applies to chlorophyll-a as well. To assess the impact of effluents of pulp industries, the accumulation of toxic substances (organochlorides) in soft tissues of mussels and muscle tissue of fish is monitored.

From literature it has been found that the Finnish-Norwegian Transboundary Water Commission launched an extensive programme in 1989 (Koskenniemi, 1990).

The programme, which includes monitoring of macroinvertebrates, investigates the general state of the river and its natural value. The sampling sites are riffles, and apart from the kick-net method, colonization methods have also been used. In the examination of the results, ordination and indices (e.g. BMWP) will be used in the programme that started in 1994.

GERMANY

Information is reported for 13 transboundary rivers. Besides information on larger rivers, such as the Rhine, Donau, Elbe and Salzach, information was also sent on smaller ones, such as the Issel, Niers, Vechte etc. Routine biological monitoring takes place in (nearly) all rivers. Rhine monitoring is part of an international programm, integrated in national and statual monitoring systems. Such an international programm also exists for the Elbe river. Strategy and choice of parameters are updated regularly. Biological structure is monitored for phytoplankton, zooplankton, invertebrate and fish communities. A saprobic index is calculated. Toxicity tests are performed with bacteria, algae and invertebrates, while accumulation of toxic substances is monitored in fish, regularly, on a 5-yearly base.

HUNGARY

The Hungarian biological monitoring programmes include some bacterial analyses and chlorophyll-a content. Sampling is performed weekly.

LATVIA

The routine biological monitoring on transboundary rivers since 1994 concerns bacteria, phytoplankton and zoobenthos. The aims are assessment of the ecological status and quality of water body, examination of biological quality of the receiving water on the transboundary hydrofront and determination of the suitability of the water body for fisheries and other uses.

Information on all three biotic groups are elaborated into hydrobiological indices which make up a water quality classification scheme together with a hydrochemical water pollution index (WPI). The microbiological index is the number of saprophytic bacteria.
The phytoplankton or periphyton (distinction is not clear in questionnaire) is evaluated in the saprobity index of Pantle & Buck. In addition to this index, biomass estimates, summation of population biomasses and determination of dominant species are also performed.

The zoobenthos (or invertebrate) community is sampled by means of a Bottomscraper and Peterson grab and evaluated by means of a biotic index (not specified) and the relative Oligocheata abundance. Species composition is determined at species level.

NETHERLANDS

On the large rivers in the Netherlands routine biological surface water monitoring is performed, but not necessarily at the border location. The objective is to get an indication of the ecological status, detect trends and test the status against standards. Information is used to detect trends in the status of the ecosystem and test the status against standards and reference or target situations. The monitoring in large rivers consists of analyses of biomass and bacteria, accumulation of toxic substances in mussels and eel, the biological structure of phytoplankton, phytobenthos, zooplankton, invertebrates, fish and birds. Macroinvertebrates are collected by means of artificial colonizing substrates.

On the smaller transboundary rivers and lowland streams, biological monitoring and assessment is performed at the border location by regional water authorities. Most regional water boards have developed their own biological assessment method, which differs from the national method for large rivers. Since 1992 a nation-wide ecological assessment method has become operational for small running waters based on macroinvertebrate community (STOWA,1994).

NORWAY

The biological variables monitored are limited and mostly orientated on biological structure of phytoplankton and zooplankton community. Periphyton is removed from natural substrates and provides algal indicators for determination of the general degree of pollution. The zoobenthos is sampled with the kicking method and Surber sampler. In some rivers, the fish community is sampled by means of electrofishing and evaluated. In one river, bioaccumulation of metals is monitored in fish, while in other river heavy metal content is monitored in water plants (Fontinalis spp).

POLAND

The Polish questionnaire explicitly reports the absence of routine biological surface water assessment of transboundary rivers. Only monitoring of coliform bacteria is reported. However, in other streams in Poland the Saprobic index based upon phytoplankton and zooplankton has until recently been in use. The calculation method of Pantle & Buck was applied, in combination of the species indicator value list of Sládecek (1973). Since 1993 only examination of bioseston is obligatory.

PORTUGAL

"Routine" biological surface water monitoring is not yet applied in transboundary rivers. Nevertheless, in some transboundary rivers and other Portuguese rivers, biological assessment has been performed, on a regular base since 1986. The biological variables monitored concern mainly phytoplankton, periphyton and zoobenthos or macroinvertebrate community. The sampling methods of Strickland & Parsons (1972) and Utermohl (1958) are used in case of phytoplankton. For the zooplankton, net-filtration is used. Periphyton is removed from artificial substrates. The zoobenthos is sampled by handnet and the fish community is sampled by means of electrofishing and gill nets.

Classification of biological water quality is based on the Belgium biotic index (see annex 3). During the hydrological year (October '94 - September '95) a biological assessment programm for rivers and reservoirs in the north of Portugal is under development, parallel with the river water quality inventory (use-related physical and chemical classification; nationwide).

ROMANIA

It is reported that routine biological surface water monitoring takes place at all rivers with a frequency of 4 times a year. Variables include biomass, bacteria, accumulation of toxic substances in mussels and fish and biological structure of phytoplankton, zooplankton, invertebrates, fish and birds. Romanian river water quality standards regard algal biomass and total coliforms as determinands of a biological kind. Planktonic biomass is used for classification of water quality into trophic zones. For phytoplankton and zooplankton, a Saprobic index according to Pantle & Buck, modified by Knöpp is applied.

With respect to the invertebrate community, Rumania reports little use of this information due to the absence of adequate equipment and methodology. The sampling method is not standard and the full range of invertebrate fauna is not sampled.
Information on submerged macrophytes and marginal vegetation are not obtained regularly.
In special studies heavy metal and pesticide accumulation in mussels and fish is monitored. Fish and bird inventories appear irregularly and in specific areas only. For fish, an indicator list for water quality is provided. Toxicity testing is performed with algae, invertebrates and fish.

SLOVAK REPUBLIC

Biological monitoring of Slovakian transboundary rivers consists in general of a number of microbiological parameters. The former consists of psychrophilic, mesophilic bacteria, faecal and total coliforms and faecal streptococci. Hydrobiological monitoring is based on community structure approach, while assessment is performed by means of the saprobic index, calculated according to Pantle & Buck, for plankton, microphytobenthon and macrozoobenthon.

The aforementioned microbiological and hydrobiological parameters are one part of five sets of parameters which make the classification scheme for water quality.

UKRAINE

Biological monitoring of Ukrain transboundary rivers in general consists of measurement of biomass and determination of biological structure of phytoplankton and zooplankton and of microbiological parameters. Saprobic indices can be calculated.

UNITED KINGDOM

The biological monitoring of the transboundary rivers between Northern Ireland and the Republic of Ireland consists in general on macrophytes and invertebrates. The biotic score results from the River Invertebrate Prediction and Classification System (see annex 4), which is used for classification of rivers.

4.3 Biological structure

A summary on the biological surface water quality monitoring and assessment methods concerning biological structure is given in table 4.1. Only application in transboundary rivers is reported. Details on sampling devices and sampling quantities have been left out. These matters are seldom reported.

Comparison of the data in table 4.1 with the state-of-the-art presented in chapter 3 leads to a number of conclusions:
- In more than half of the countries some kind of biological assessment regarding biological structure is performed;
- macroinvertebrate community structure is most often used in monitoring and assessment, followed by phytoplankton;
- the dominant biological assessment method appears to be a saprobic system with the aid of the saprobic or saprobity index. The wide-spread use of a biotic index of biotic score that has been pointed out in chapter 3 (section 3.4) is not reflected by the reporting countries. Biotic scores are either not applied in these countries or are not applied in transboun-

dary rivers. Combining literature sources and results of the enquiry, there appears to be a general geographical division in which the biotic index is more often used in Western-European countries and the saprobic index is more often used in Central and Eastern-European countries.

- The main impact on river water quality that is being assessed in the water body is the saprobic state or influence of discharged domestic waste water.
- Routine biological river quality assessment is focused upon the biotic communities of the water body or aquatic zone, while amphibious and terrestrial zones are hardly involved. The little attention for higher trophic levels like fish, birds and mammals in rivers and riverine ecosystems might be an underestimation of current practice. Traditionally these groups have been of interest and have been monitored by nature conservation institutions rather than water quality management.
- While realising that the questionnaire did not contain any specific questions on habitat quality or ecological assessment, there appears to be no current practice on this issue in transboundary rivers.
- The enquiry did not contain specific questions on classification schemes and presentation methods for biological assessment methods. The reported current state is therefore not complete.

Table 4.1
Current practice on the use of biological structural aspects of transboundary river ecosystems in ECE countries.

notes:
P&B = Pantle & Buck;
x = present; frequency not specified
+ = present
# = 1/4 = once per 4 years
= bioseston = plankton (=phytoplankton, zooplankton, mycoplankton, bacterioplankton), microphytobenthon, macrozoobenthon.
= microphytobenthon
biotic groups: PP= phytoplankton, PB = phytobenthos/periphyton, MP = macrophytes, ZP = zooplankton, M = macroinvertebrates (or macrozoobenthon), F = fish, B = birds.

	name of method (calculation method)	**frequency of monitoring biotic groups (number per year)**							**use in classification**	**remarks**
		PP	**PB**	**MP**	**ZP**	**M**	**F**	**B**		
Austria	biocoenotic analysis (saprobic index)	1	1-2	1	1	1-2			+	
Bulgaria	(under development) (biotic & saprobic index)					x				
Croatia	Saprobity index (P&B/Sládecek)	2	2		2	2			+	
Czech	biocoenotic analysis	1/3#	1/3#	1/3#		1/3#	1/3			Elbe river.
Republic	saprobic index	12##				1/5#	#		+	national network
Estonia	no biological assessment									
Finland	no biological assessment									
Germany	saprobic index	x			x	x	x		+ from weekly	frequency differs (phytoplankton) to yearly (macrobenthos)
Hungary	saprobic index	12	12		4		2/3 #			
Latvia	saprobity index (from 1994)		4			from 1995			+	
Netherlands		1/4 #	1/4	1/4	1/4	1/4	1	1		in small rivers different systems
Norway		0-2		0-1	first time 1993	first time 1993	1-2			
Poland	no biological assessment of transboundary rivers	x								saprobic index is applied in other rivers
Portugal	no yet applied									
Romania	saprobic index (P&B/Knöpp)	4			4	4	1	1/2		
Slovak Republic	saprobic index of bioseston# (P&B/Sládecek)	12-24	4##		12-24	4-6			+	
Ukrain	saprobic index	3-4	3-4			3-4			+	
United Kingdom	RIVPACS			1		3			+	

4.4 Functional and microbiological parameters

Table 4.2. provides a summary on reported and applied methods with respect to ecosystem functioning in the sense of occurring processes and microbiological analyses and assessment. The numbers in the table represent (ranges of) applied sampling frequencies.

It is apparent from table 4.4. that both application and frequency of ecosystem functioning and microbiological variables show a large variation for the reporting countries. Only chlorophyll-a is frequently used as a variable providing information on ecosystem functioning. The value of chlorophyll-a lies mainly in the insight in the degree of eutrophication of the system. In

Table 4.2
Current practice on the use of functional aspects of transboundary river ecosystems and microbiological assessment in ECE countries.
membrane filtration
x present, frequency not specified
Numbers in table refer to frequency per year; ranges refer to different waters.
chl-a = chlorophyll-a content
Faecal coliforms = thermo-tolerant bacteria of coligroup (=E.coli)
Total coliforms = E.coli, Enterobacter sp, Klebsiella sp, Citrobacter sp)
1) special project on Elbe river
2) introduced in standard network from 1994.

	functional parameters			microbiological parameters				
	biomass	chl-a	primary production	faecal coliforms	total coliforms	faecal streptococci	Salmonella	other
Austria		12		4-12	4-12	4-12		
Bulgaria								
Croatia								MPN37
Czech Republic	1/3y[1)]	(12)[2)]		12	12	12		mesophil.bact.
Estonia								
Finland				3-12		4-12		
Germany	12	52	52	13	13			colony count 22°C
Hungary		52		52		52	52	
Latvia	x	x		x	x	x		coliphages, heterotrophic plate count, index of saprophyt bacteria
Netherlands		26		13			13	
Norway				12(44°C)#		12(37°C)		# membrane filtration
Poland		2-4	2	26				
Portugal								
Romania	4	4		4	4	4	4	mesophil bacteria
Slovak Republic	12-24	12-24		12-24	12-24	12-24		psychrophilic, mesophylic bacteria
Ukrain	4			12	12			coliphages, saprophyt bacteria.
United Kingdom								

some countries chlorophyll-a content is one of the water quality criteria in a classification scheme e.g. in Finland. Biomass and primary production are rarely used in routine biological surface water monitoring in transboundary rivers.

The assessment of hygienic state of river water by means of microbiological methods is applied in almost all reporting countries. The assessment of faecal coliforms is the most widely used method, immediately followed in ranking by the faecal Streptococci. Less frequently used are total coliforms, total bacterial counts and Salmonella. Only very limited information can be deducted from questionnaires with respect to methodology. Sampling frequencies vary from quarterly to weekly.

4.5 Toxicity, mutagenicity and bioaccumulation

Table 4.3. summarizes the methods reported by the ECE countries for routine monitoring of toxicity, mutagenicity and accumulation in transboundary rivers.
From this table it can be concluded that assessment of impact of toxic substances in river water and bottom has found limited application so far in transboundary waters. The impression arises that this is due to a certain historical separation of disciplines (hydrobiology and ecotoxicology) rather than a result of a lack of available methods. In recent years these two disciplines have tended to converge. Researchers concerned with biological assessment of running waters call for methods to evaluate toxicological effects on stream communities, while ecotoxicologists are aware of the

limitations of laboratory bioassays e.g. in risk assessment of new compounds (Friedrich, 1992; Van Leeuwen, 1995). Bioaccumulation methods are likewise rarely used.

Table 4.3
Current practice on assessment of toxicity, mutagenicity and bioaccumulation of transboundary rivers in ECE countries.
1) in special projects (Elbe and Oder)

	toxicity		**mutagenicity**	**bioaccumulation**
	river water	**sediment**	**Ames**	
Austria				
Bulgaria				
Croatia				
Czech Republic	Daphnia, fish		(occasionally)	fish, macroinvertebrates, macrophytes 1/3 y 1)
Estonia				
Finland		chironomids, oligochaetes		
Germany	bacteria,algae,invertebrates,fish.			fish (1/5y)
Hungary				
Latvia	bacteria,protozoa,invertebrates	invertebrate (T. piriformis)		
Netherlands	invertebrates (porewater)	1/4 chironomids		mussels,
fish 1/4 y				
Norway				fish (1-2), water plants
Poland	Lebistes reticulatus? (occasionally)			
Portugal				
Romania	invertebrate,fish			
Slovak Republic				
Ukrain*				
United Kingdom*				

5. Recommendations for harmonisation

General considerations on biological monitoring and assessment

It is generally accepted in literature that biological monitoring and assessment provides much extra value to the 'traditional' chemical monitoring of river water quality. Selection of suitable biological methods should regard and be related to the assigned functions of the river and their respective objectives.

Monitoring and assessment of the ecological status of a river, which represents an intrinsic natural value rather than an human assigned functional use, can only be performed by means of biological or ecological methods.

From the reviewed state-of-the-art (chapter 3) it can be concluded that there is a great variety of available biological assessment methods both in number and in scope of application. The scope of the methods ranges from assessment of a single impact on a single compartment of the river to assessment of multiple aspects of the entire riverine ecosystem. A general observation is that the number of comparable available methods decreases as the scope of the method increases.

From literature sources, a development from chemical and biological to ecological assessment can be distinguished. Nevertheless there is at present no 'holistic' method which covers all kind of potential impacts at the level of a riverine system or catchment area. Reviewing literature, many calls can be found for an integrated approach in which combined application and evaluation of specific methods for specific problems provide better and more comprehensive insight in river water quality.

Biological assessment tools should be carefully chosen with respect to designated functional uses and/or the intrinsic ecological value of the riverine ecosystem

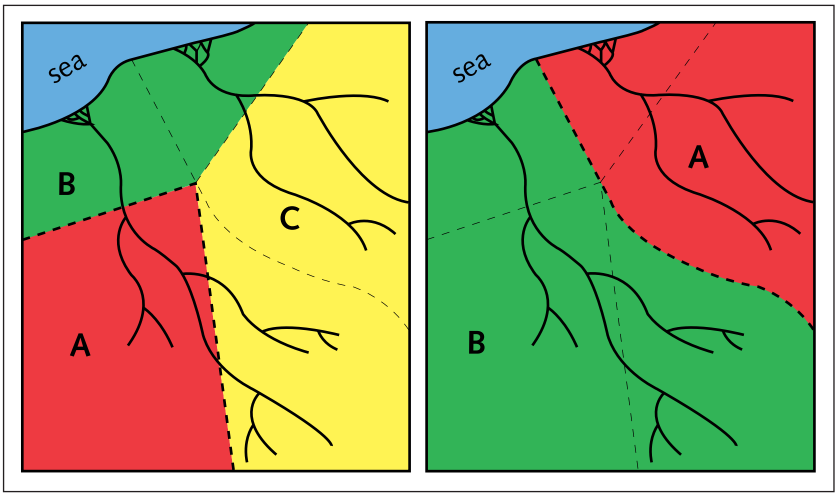
Recommendations for harmonisation of current practices in UN/ECE Task Force countries

Evaluation of the current practices on biological assessment in ECE Task Force countries (chapter 4) reveals great differences in the extent of implementation and use of biological methods in the countries involved. Only a few methods are commonly applied.

This report takes as a starting-point that recommendations for harmonisation should join with current practices, while considering the perspective of future developments, rather than be restricted to application of the state-of-art methods only. Application of new monitoring and assessment techniques requires implementation time, building of knowledge and sufficient financial and administrative means.

River quality assessment should be harmonised over ecological relevant borders of catchment areas rather than along political borders (as illustrated by figure 5).

On the issues of monitoring strategy aspects like site selection, sampling frequencies and sampling methodology, the information in the enquiries shows a large variation in the level of detail, hindering balanced recommendations for harmonisation. Sampling frequency and methodology is closely related to the biotic group that is concerned.



Figuur 5.1
River assessment strategies.
Left: present: assessment methods and river management along political borders.
Right: future situation: management along catchment borders.

Recommendations on specific methods are as follows:

> The benthic macroinvertebrate community is considered as a good practical tool for routine monitoring and assessment of biological quality of the aquatic zone of rivers. Determination on species level is essential.

biotic index
The methodology of a biotic index or score can be used in one modification or another over a very wide geographical area. Regional differentiation is however a necessary but possible prerequisite. Regions should be based on ecological borders rather than political borders. The family level identification makes determination rapid and prevents the method from being too much differentiated. The development of RIVPACS and the calculation of the Ecological Quality Index (EQI) shows that the method can provide measures that can be implemented in standards.

Implementation of this methodology over a wide area requires the need for well defined reference situations and communities in all different river types over the countries involved.

> It is recommended in assessment of biological river water quality to use a biotic index based on macroinvertebrates whether or not regionally differentiated. The establishment of a database of well defined unaffected reference communities based on ecoregions will facilitate the use of implementing biotic index in setting water quality standards. Considering the future information needs, a determination on family level 'as a rule' will be not enough.

saprobic index
The saprobic index is the most commonly used biological assessment method in the reporting countries in the assessment of biological status or quality of river water.

The purpose of this index is to classify the saprobic state of running waters, covering the full range from unpolluted to extremely polluted waters.
Currently, two formulas are mainly in use to calculate the saprobic or saprobity index, being the formulas according to Pantle & Buck and according to Zelinka & Marvan. Although the former is more frequently used, the formula of Zelinka & Marvan is preferred on theoretical grounds. It takes the distribution of indicator species over several saprobic zones into account, whereas the Pantle & Buck formula considers each species to belong to one zone only. The present state of data automation with computers eliminates the major drawback of the Zelinka & Marvan formula being the elaborated calculation. The use of relative abundances or estimates reduces the effort of identification of species in the sample.

Most countries use the species indicator value list of Sládecek, which dates from 1973. The latest revision of this list which has been put forward by Germany is recommended to use for benthic invertebrates. The evaluation of one biotic group provides sufficient information on saprobity.

It is recommended in assessing the saprobic state of a river, to use the saprobity index according to Zelinka & Marvan combined with the most recent species indication value list. In case of using the saprobity index the limitations have to be taken into account.

Sampling frequencies and observed biotic group show important differences. For upstream courses, macroinvertebrates are preferred and for downstream courses of large rivers phytoplankton may serve the best practical means.

Recommendations on future developments

Adapting new monitoring and assessment methods in current practices can be performed by a step by step extension, knowing that at present no integrated, comprehensive method is available. A variety of measures are required to adequately assess river quality. It has to be understood that (current practices on) chemical monitoring serve the objectives related to a number of functional uses in a satisfactory way. The assessment of ecological status and quality however could be extended.

The present biological assessment methods based on the macroinvertebrate or phytoplankton community composition, which basically involve organic pollution, should be extended with other aspects of the river water body like habitat quality, toxicity and sediment quality. On habitat quality a number of regional methods are under development. Toxicological and sediment quality assessment methods are sufficiently available from the research field of ecotoxicology, mostly involving experimental or laboratory setups.

In the perspective view of a river as part of a riverine ecosystem with an intrinsic ecological value, other biotic groups like amphibians, water birds and mammals in other compartments like banks, marshes and floodplains have to be considered. These aspects have to be made measurable.

It is recommended to extend chemical monitoring and assessment to ecological assessment. This can be achieved by additional successive steps:
- **assessment of biological status** of the river water body with respect to biotic community composition, structure and functioning;
- involving the **assessment of abiotic factors or habitat quality** in relation to biotic communities, ecological assessment;
- **application of ecotoxicological tools** like experimental and laboratory setups;
- extension of the assessment of river water body to the other zones of the riverine ecosystem; **an ecosystem approach.**

It appears to be becoming common practice to evaluate the present observed state on a specific element or aspect against a reference or target situation. This requires a scientifically valid reference as has been developed with RIVPACS. A method in which manyfold aspects can be quantitatively evaluated and presented has been made available in the AMOEBA approach.

It is not to be expected nor wanted to develop complex integrated assessment methods that include all biotic and abiotic variables. Instead, such an integrated method should be composed of selected, “smart” variables, that are proven to be representative for a community, sensitive for general of specific impacts on riverine ecosystem elements.

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Microbiological methods

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Water quality - Enumeration of viable micro-organisms - Colony count by inoculation in or on a nutrient agar culture medium, 2 p.

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Water quality - Detection and enumeration of the spores of sulfite-reducing anaerobes (clostridia). Part 1: Method by enrichment in a liquid medium, 3 p.

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ISO 7899-1, 1984.
Water quality - Detection and enumeration of faecal streptococci. Part 1: Method by enrichment in a liquid medium, 3 p.

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ISO 8199, 1988.
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Water quality - Detection and enumeration of *Pseudomonas aeruginosa.* Part 1: Method by enrichment in liquid medium, 5 p.

ISO 8360-2, 1988.
Water quality - Detection and enumeration of *Pseudomonas aeruginosa.* Part 2: Membrane filtration method, 5 p.

ISO 9308-1, 1990.
Water quality - Detection and enumeration of coliform organisms, thermotolerant coliform organisms and presumptive *Escherichia coli.* Part 1: Membrane filtration method, 10 p.

ISO 9308-2, 1990.
Water quality - Detection and enumeration of coliform organisms, thermotolerant coliform organisms and presumptive *Escherichia coli.* Part 2: Multiple tube (most probable number) method, 9 p.

ISO 9998, 1991.
Water quality - Practices for evaluating and controlling microbiological colony count media used in water quality tests, 22 p.

Sampling and analysis

ISO 5667-1, 1980.
Water quality - Sampling. Part 1: Guidance on the design of sampling programmes, 13 p.

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Water quality - Sampling in deep waters for macro-invertebrates - Guidance on the use of colonization, qualitative and quantitative samplers, 13 p.

ISO 10260, 1992.
Water quality - Measurement of biochemical parameters - Spectrometric determination of the chlorophyll-a concentration, 6 p.

Toxicological methods

ISO 6341, 1989.
Water quality - Determination of the inhibition of the mobility of *Daphnia magna* Straus (Cladocera, Crustacea), 7 p.

ISO 7346-1, 1984.
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ISO 7346-2, 1984.
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ISO 7346-3, 1984.
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Annex 1. UN/ECE-countries and involvement with Helsinki-Convention (1992)

	ECE-country signed	Convention	In Task Force Mon. & Ass.
Albania	x	x	
Andorra	x		
Austria	x	x	x
Azerbaidjan	x		
Belgium	x	x	
Belarus	x		x
Bosnia and Hercegovina	x		
Bulgaria	x	x	x
Canada	x		
Croatia	x	x	x
Cyprus	x		
Czech Republic	x	x	x
Denmark	x	x	
Estonia	x	x	x
European Union	x	x	
Finland	x	x	x
Former Yugoslavian Republ. of Macedonia	x		
France	x	x	
Georgia	x		
Germany	x	x	x
Greece	x	x	x
Hungary	x	x	x
Iceland	x		
Ireland	x		
Israel	x		
Italy	x	x	
Kazakhstan	x		
Kirgizistan	x		
Latvia	x		x
Lichtenstein	x		
Lithuania	x	x	
Luxembourg	x	x	
Malta	x		
Moldova	x	x	
Monaco	x		
Netherlands	x	x	x
Norway	x	x	
Poland	x	x	x
Portugal	x	x	x
Romania	x	x	
Russian Federation	x	x	x
San Marino	x		
Slovak Republic	x		x
Spain	x	x	
Sweden	x	x	
Switzerland	x	x	
Turkey	x		
Turkmenistan	x		
Ukraine	x		x
United States of America	x		
Uzbekistan	x		
United Kingdom	x	x	x

Annex 2. Diversity indices and comparative indices

(reprinted from Hellawell, 1986 and Boyle *et al.*, 1994)

Diversity indices

1. William's Alpha index (Fisher *et al.*, 1943):

$$S = \log_e N / \alpha$$

where S = no. of species in community
N = no. individuals in community
α = index of diversity

2. Diversity index (Menhinick, 1964)

$$\text{diversity index } I = \frac{S}{\sqrt{N}}$$

symbols as above

3. Information theory index (Shannon, 1948):

$$H = -1 * \Sigma \left(\frac{n_j}{I}\right) * \ln \left(\frac{n_j}{I}\right)$$

where H = homogenity
I = total no. of individuals in community
n_j = no. of individuals of j-th species

4. Brillouin's H:

$$H = \frac{1}{I} [\ln (I!) - \Sigma \ln (n_j!)]$$

symbols as above;
ln(I)! = natural logarithm approximated using Stirlings formula

5. Diversity index (Simpson, 1949):

$$\text{diversity index } I = \Sigma \frac{n_j[n_j -1]}{N (N-1)}$$

symbols as above

6. Diversity index (Margalef, 1961):

$$D = \frac{S-1}{\ln (I)}$$

symbols as above

7. Diversity index (McIntosh, 1967):

$$\text{index } I = 1-\sqrt{\Sigma n_j^2}$$

symbols as above

Comparative indices

8. Jaccard's index:

$$\text{Jaccard's Index} = 100 * \frac{S_c}{S_i + S_j}$$

where S_c = no. of species in common between two communities
S_i, S_j = no. of species in communities i,j

9. Quotient of similarity (Sorensen, 1948):

$$I = \frac{2S_c}{(S_i + S_j)}$$

symbols as above

10. Percent Similarity (PCS):

$$PCS = 100 * [1.0 - 0.5 \sum |p_{oj} - p_j|]$$

where $p_j = n_j/I_o$ = proportion of perturbed community belonging to species j
$p_{oj} = n_o j/I_o$ = proportion of original community belonging to species j

11. Pinkham & Pearson:

$$B = \sum \frac{\text{ratio (j)}}{S_o}$$

where S_o = total no. of species in original community
ratio (j) = $\min[n_j, n_{oj}] / \max[n_j, n_{oj}]$

Annex 3. Belgian biotic index

In Belgium, a biotic index is in use for routine monitoring and assessment of running waters on a nationwide scale (Vlaamse Milieumaatschappij, 1994). The system will be presented here as an example of the assessment methods group of biotic indices and biotic scores (see Section 3.4.).

Objective

The method aims at the biological quality assessment of running waters in Belgium.

Principle

The Belgian Biotic Index (BBI) has been deducted from the first biotic index method (Trent Biotic Index, Woodiwiss, 1964) and the biotic index proposed by Tuffery & Verneaux (1968); De Pauw & Vanhoren, 1983; NBN, 1984; De Pauw & Vannevel, 1990).

Execution of the method concerns the following steps: sampling of macroinvertebrate community, identification and calculation of the Belgian Biotic Index. The calculation is performed by using the table with indicating faunistic groups and number of systematic units. A systematic unit involves mostly taxonomical groups at genus or family level. The resulting value of the Belgian Biotic Index is classified on a 5-class quality scale ranging from lightly polluted or unpolluted to very heavily polluted.

Table A1
Calculation table for the Belgian Biotic Index.
[1] S.U.: number of systematic units observed of this faunistic group.

I Faunistic group	**II**	**III Total number of systematic units present**				
		0-1	**2-5**	**6-10**	**11-15**	**16 and more**
		Biotic Index:				
1. Plecoptera or Ecdyonuridae	1: several S.U.[1]	-	7	8	9	10
	2: only 1 S.U.	5	6	7	8	9
2. Cased Trichoptera	1: seweral S.U.	-	6	7	8	9
	2: only 1 S.U.	5	5	6	7	8
3. Ancylidae or Ephemeroptera (exceqt Ecdyonuridae)	1: more than 2 S.U.	-	5	6	7	8
	2: 2 or < 2 S.U.	3	4	5	6	7
4. Aphelocheirus or Odonata or Gammaridae or Mollusca (except Sphaeridae)	0: all S.U. mentioned above are absent	3	4	5	6	7
5. Asellus or Hirudinea or Sphaeridae or Hemiptera (except Aphelocheirus)	2	3	4	5	-	-
6. Tubificidae or Chironomidae ot the thummi-plumosus group	1	2	3	-	-	-
7. Eristalinae (Syrphidae)	0: alle S.U. mentioned above are absent	0	1	1	-	-

Scope of application

The BBI has been designed for use in Belgium. The running waters of Belgium range from shallow, slow to fast running waters to deep watercourses.

Information requirements

Qualitative collecting of macroinvertebrates is performed by a hand-net in all accessible micro-habitats during a certain time: 3-5 minutes. The sampled organisms are identified at the family or genus level, depending on the order concerned.

The genus level is applied to Plathelmintes, Hirudinea, Mollusca, Plectoptera, Ephemeroptera, Odonata, Megaloptera, Hemiptera wheras the family level is applied to Oligochaeta, Crustacea, Trichoptera,Coleoptera, Diptera, Chironomidae thummi-plumosus or Chironomiade non-thummi-plumosus. Every observed genus or family represents a systematic unit. After identification, the presence of the most sensitive faunistic groups (column I) and the number of systematic units of a particular group (column II) as well as the total number of systematic units (colomn III) present in the sample is counted. From a table, the combination of both variables results in a biotic index.

A systematic unit represented by a single individual is not taken into account because its occurrence may be accidental. In deep and large rivers colonizing substrates may be applied (De Pauw *et al.*,1993).

Presentation

The results of the biotic index are classified on a quality scale, provided with a colour banding.

Table A2
Classification and colour coding of biological assessment results in Belgium.

Class	Biotic Index	Significance	colour
I	10-9	lightly or unpolluted	blue
II	8-7	slightly polluted	green
III	6-5	moderately polluted -critical situation	yellow
IV	4-3	heavily polluted	orange
V	2-1	very heavily polluted	red
	0	absence of macroinvertebrates	black

Annex 4. RIVPACS (River InVertebrate Prediction and Classification System)

As a result of a nationwide research programme on the macroinvertebrate communities of British rivers in the years 1977-1988, the Freshwater Biological Association has developed an alternative system for biological assessment of river quality (Wright *et al.*, 1988, 1989, 1993). At the time, biological surveillance of UK rivers was performed by means of the BMWP score (Biological Monitoring Working Party; Chester, 1980) and the ASPT (Average Score Per Taxon; Armitage *et al*.,1983), which can be considered members of the group of biotic indices and biotic scores (see Section 3.4.).

The newly developed system, RIVPACS, has been used in the nationwide biological assessment of rivers in the United Kingdom in 1990. By means of cluster analysis of a large set of ecological data from unpolluted references rivers in the UK, a classification scheme was developed. Afterwards, a multiple discriminant analysis was applied as a prediction technique.

Approach

The approach of RIVPACS comprises four major steps: measurement of a number of chemical and/or physical features of a river site; prediction of macroinvertebrate community in terms of probability of presence at the family level; sampling and identification of macroinvertebrate community at the site; and evaluation of degree of disturbance by comparison of observed and predicted number of taxa or index score (ASPT or BMWP). The predicted community (score) is a site-specific assessment endpoint. The endpoint predicted can also indicate the natural range of variation that might expected at each site due to random sampling error.

Scope of application

The scope of application is at the moment restricted to the United Kingdom due to differences in occurrence of species and ranges of enviroomental variables (like latitude and longitude) between the United Kingdom and other countries. Some testing experience is available in Spain, Canada and Australia. The basic approach and multivariate techniques are portable to other nations.

The essenuial requiremeots for developing the RIVPACS approach in other regions in Europe are the availability of a wide range of good quality streams and rivers to act as reference sites, coupled with use of standard sampling techniques, a uniform level of identification and access to good quality environmental data much of which may be map-based.

Information requirements

The measurement of 10 to 12 different environmental variables, which are grouped into six options, is required at a site under study.

Sampling is performed by a pond-net in all major habitats, in proportion of occurrence, using kicking and sweep-netting for 3 minutes. Data from three seasons are required. Identification of macroinvertebrate species is performed at (BMWP) family level. In the original dataset, the identification level was at species level. It is possible to predict species probability of occurrence and so calculate indices other than BMWP.

Table A3
The six environmental options available for predictions in RIVPACS II.

Eight variables common to all menu options:

Distance from source	Mean water width
Mean substratum	Mean water depth
Altitude	Latitude
Discharge category	Longitude

plus the following, according to option:

option	1	2	3	4	5	6
alkalinity	+	+		+		+
Slope	+		+	+		+
Mean air temperature	+	+	+		+	
Air temperature range	+	+	+		+	
Chloride						+

With the aid of the abiotic analyses, an ‘expected’ biotic score is calculated. The 'observed' biotic score is calculated on the basis of the sampled community. Afterwards, the relation between observed and predicted provides a measure, called Ecological Quality Index (EQI), which could be classified into four quality classes:

Table A4
Biological banding of ASPT, number of taxa and BMWP (3 EQI's) score based on sampling in three seasons.

biological class	Obs/exp. ASPT	Obs/exp. no.taxa	Obs/exp. BMWP score
A (highest)	≥ 0.89	≥ 0.79	≥ 0.75
B	0.77-0.88	0.58-0.78	0.50-0.74
C	0.66-0.76	0.37-0.57	0.25-0.49
D	<0.66	<0.37	<0.25

One of these EQI's could be used in setting statutory water quality objectives (Seager, 1993).

Annex 5. Ecological assessment for running waters in Germany

Saprobic system for water quality

In (the Former Federal Republic of) Germany the saprobic system (Saprobiensystem) has been in use for routine monitoring of running waters since the late seventies at the federal and state level (LAWA, 1976; LWA, 1982). During the first decade, the existing species indicator list was used (Sládecek,1973). Recently the species list was revised by a group of experts with the aid of statistical analysis of long term monitoring data of water quality (Friedrich,1990). This revised list has become a part of a German standard (DIN 38410) and is limited to benthic macroinvertebrates. The calculation of the saprobic index is based on the formula of Zelinka & Marvan (1961) (see Section 3.5). The results, classified into 7 water quality grades, were presented in water quality maps (Gewässergütekarte) in which stretches of running waters are coloured. Furthermore, the saprobic index became a part of the General Quality Requirement for running waters for use in water management plans, e.g. in Nordrhein-Westfalen. The saprobic system has proven to be valuable in assessing the biological water quality (Friedrich, 1992).

Ecological approach: structure quality

In recent years, after significant reduction of the load of organic, biodegradable substances and successive improvement of biological water quality, the awareness of the deficits in the structure and functioning of running waters arose. Currently an ecological assessment method is under development which in the near future has to lead to a federal Water Quality Atlas. This atlas will contain the following elements for running waters:
- water quality map based on the saprobic index, translated into water quality classes. It should be noted that the class coding is not a rigorous scheme, but results after careful examination of all ecological information available (Friedrich, pers.comm.,1995);
- stream structure quality map: this method is at the testing stage now (see below).
- mapping of some chemical features;
- mapping of acidity of small running waters. This method has been tested. (Steinberg & Putz).

The stream structure assessment is made for three zones: the aquatic, riparian (banks) and the terrestrial zone or river valley. 27 single parameters, grouped into 6 main parameters, concerning structure are distinguished.

Scope and application

The stream structure assessment can be applied for a range of medium size running waters from head stream to small, fordable rivers. The underlying typology of running waters contains 11 types. The assessment method is now under development in the state of Nordrhein-Westfalen in Germany but will be applied throughout Germany in the near future after the testing phase. efforts have also been made in assessing the structure quality in other German states (Wild, 1992).

t has to be noted that the ecological assessment method is concerned about abiotic structural features of running waters that are important to the ecosystem, but the method can not be considered a holistic ecological assessment method (Friedrich,1993). The assessment of water quality and structure quality has been kept separate on purpose. Thus, the assessment is prevented from being complex due to many interactions between biotic and abiotic factors. Furthermore, the structure assessment allows deficits to be made immediately clear to daily water management of organization and maintenance. Instruments and measures for improvement or rehabilitation result directly from the assessment.

Information requirements

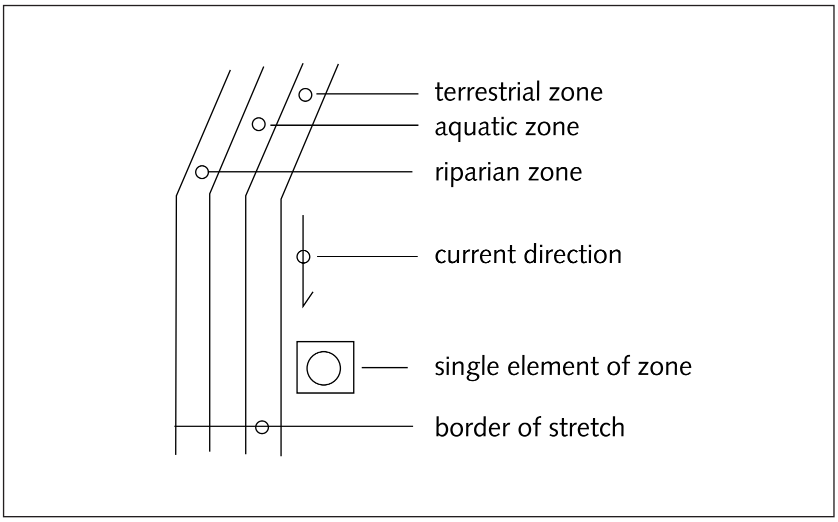
Monitoring structural quality is performed by means of standardised protocols and forms to be filled out in the field. The Starting point for the assessment is the natural reference situation (Leitbeild) of 6 main parameters for the water under investigation. Knowing the water type and respective reference situation, the field worker can estimate the deviation of the site under study and classify for the 6 main variables by (grouped) averaging of all 27 structure variables. The assessment is made for stretches which differ for one or more variables, with a maximum length of one kilometer.

Presentation method

The resulting classes for the main variables are averaged for the respective zones: aquatic, riparian and terrestrial zone. For the ecological assessment a cartographical lay out has been developed (figure A1). The colour coding is as follows:

ecological stream structure class	degree of impairment	colour
1	virtually no impact (kaum beeintrachtigt)	dark blue
2	little impact (gering beeintrachtigt)	light blue
3	medium impact (massig beeintrachtigt)	dark green
4	clear impact (deutlich beeintrachtigt)	light green
5	weak impairment (merklich geschadigt)	yellow
6	strong impairment (stark geschadigt)	orange
7	severe impairment (ubermaßig geschadigt)	red

Figure A1
Cartographic legend for water structure quality in Germany.



It should be noted that the ecological structure assessment for the aquatic zone is not represented by the saprobic index, but on structural variables like longitudinal structure, curves etc. The saprobic index is used to assess the biological water quality and is presented on separate maps.

Annex 6. Ecological assessment method for Dutch running waters (STOWA-method)

In 1992, the Foundation for Applied Research on Water Management (STOWA) published the first assessment method based on macroinvertebrates for running waters in the Netherlands, that can be considered to be of an ecological type (Roos *et al.*, 1991; STOWA, 1992).

Approach

In contrast with biological assessment methods that were developed earlier for Dutch streams and regulated streams (Moller Pillot, 1972; Tolkamp, 1985; STORA, 1988) which were based on biological variables only, the STOWA-method has been based on biological and physico-chemical variables as well as environmental and management variables (like type of maintenance, (hydro)morphology, land-use, watermanagement) using multivariate analysis techniques. The large set of existing data on water quality variables of disturbed and undisturbed locations provided the basic information for method development. These data for routine monitoring purposes were collected by local water authority boards in The Netherlands during 1980-1988.

For Dutch streams, a typology scheme of 6 types of running waters has been put forward. Yardsticks have been established for different aspects or preferences of the macroinvertebrate communities like current, saprobity, trophic state, sand, sediment/deposits, vegetation and three functional feeding groups :scrapers, grazers and deposit feeders. Examination of macroinvertebrate community yields a score on each yardstick. Afterwards yardstick scores are compared with a 5-class quality scale.

The underlying basis of the yardsticks is the evaluation of 'least' polluted and not regulated sites (in virtually total absence of natural reference sites) combined with expert opinion and literature references with autecological information. Thus the reference state is a virtual or abstract one.

Scope of application

The STOWA method is applicable for all Dutch running waters, ranging from upstream parts of hill streams (maximum altitude 300 m) to small rivers and regulated lowland streams. Following the same approach, ecological assessment methods were developed for shallow lakes, ditches, canals and stratifying lakes. In the National Aquatic Outlook in Dutch water management, the STOWA method has been adapted for regional running waters, whereas the AMOEBA-approach will be applied for the main rivers like Rhine and Meuse (see section 3.8).

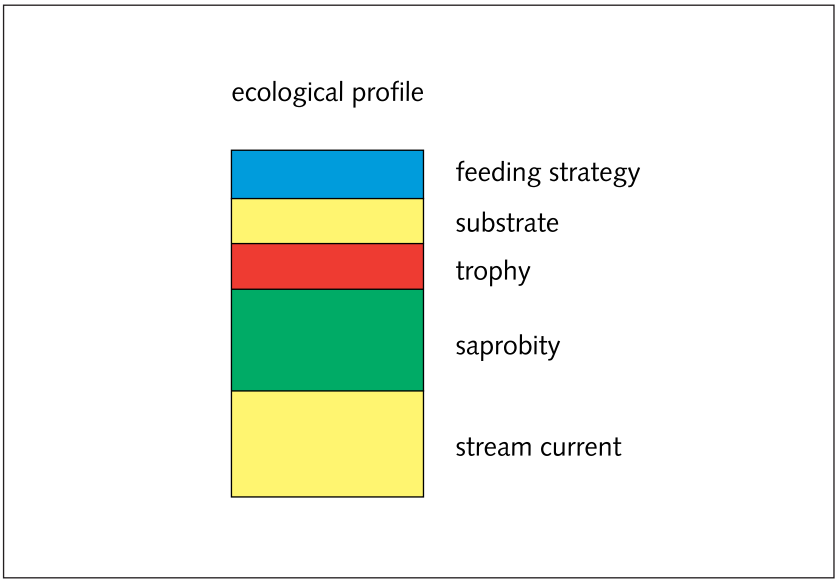
Information requirements

Application of the method requires one or two macroinvertebrate sampling events yearly, in spring and/or autumn. The advised sampling quantity is a stretch of 5 meter using a standardised (30 cm wide) hand-net in all microhabitats present. The most common level of identification is family level, but in some orders genus or species level has to be reached. No additional chemical sampling and analysis or collecting of environmental data is required. The 'ecological' component of the method lies in the implicitly implemented abiotic factors rather than in evaluation of abiotic variables. Furthermore, the assessment endpoints indicate which abiotic factor are most disturbing for macroinvertebrate community.

Presentation method

The method results in its comprised form in five quality levels ranging from below-lowest quality level to highest ecological quality level. The assessment method results in distinct quality levels for 5 different (aggregated) ecological aspects, namely: velocity, saprobity, trophy, substrate and feeding strategy. This is graphically constructed to give an 'ecological profile'.

Figure A2
Presentation of ecological.



The colour coding is as follows (modified from original STOWA-report for recent use in a national water quality survey):

colour	**quality level**
dark blue	highest quality level
light blue	almost highest quality level
green	middle quality level
yellow	lowest quality level
red	below lowest quality level