Hering, Daniel; Borja, Angel; Carstensen, Jacob; **Carvalho, Laurence**; Elliott, Mike; Feld, Christian K.; Heiskanen, Anna-Stiina; Johnson, Richard K.; Moe, Jannicke; Pont, Didier; Solheim, Anne Lyche; van de Bund, Wouter. 2010 The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future. *Science of the Total Environment*, 408. 4007-4019. 10.1016/j.scitotenv.2010.05.031

The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future

Daniel Hering[1], Angel Borja[2], Jacob Carstensen[3], Laurence Carvalho[4], Mike Elliott[5], Christian K. Feld[1], Anna-Stiina Heiskanen[6], Richard K. Johnson[7], Jannicke Moe[8], Didier Pont[9], Anne Lyche Solheim[8], Wouter van de Bund[10]

[1] University of Duisburg Essen, Department of Applied Zoology / Hydrobiology, D-45117 Essen, Germany

[2] AZTI-Tecnalia; Marine Research Division; Herrera Kaia, Portualdea s/n; E-20110 Pasaia, Spain

[3] Aarhus University, National Environmental Research Institute, Department of Marine Ecology, DK-4000 Roskilde, Denmark

[4] Centre for Ecology & Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, UK

[5] Institute of Estuarine & Coastal Studies, University of Hull, Hull, HU6 7RX, United Kingdom

[6] Finnish Environment Institute, Marine Research Centre, P.O. 140, FI-00251 Helsinki, Finland

[7] Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences, P.O. Box 7050, SE-750 07 Uppsala, Sweden

[8] Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, 0349 Oslo, Norway

[9] Cemagref, UR HBAN, Parc de Tourvoie, BP 44, F-92163 Antony Cedex, France.

[10] European Commission, Joint Research Centre, Institute for Environment and Sustainability, Via E. Fermi 2749, I-21027 Ispra, VA, Italy

Corresponding author: Daniel Hering, phone: ++49 201 183 3084, fax: ++49 201 183 4442, daniel.hering@uni-due.de

Correspondence address for proofs: Daniel Hering, University of Duisburg Essen, Department of Applied Zoology / Hydrobiology, D-45117 Essen, Germany; daniel.hering@uni-due.de

Abstract

The European Water Framework Directive (WFD), which was adopted in 2000, changed water management in all member states of the European Union fundamentally, putting aquatic ecology at the base of management decisions. Here we review the successes and problems encountered with implementation of the WFD over the past 10 years and provide recommendations to further improve the implementation process. We particularly address three fields: (i) the development of assessment methods (including reference conditions, typologies and intercalibration); (ii) the implementation of assessment systems in monitoring programmes; and (iii) the consequences for river basin management plans (such as the design, monitoring and success of restoration measures).

The development of assessment methods has been a transparent process and has resulted in improved and more standardised tools for assessing water bodies across Europe. The process has been more time consuming, and methods are more complex, than originally expected. Future challenges still remain, including the estimation of uncertainty of assessment results and a revision of rules in combining the results obtained with different Biological Quality Elements.

A huge amount of monitoring data is now being generated for WFD purposes. Monitoring data are not centrally stored and thus poorly accessible for purposes beyond the WFD. Future challenges include enhanced data accessibility and the establishment of a Europe-wide central monitoring network of reference sites.

The WFD River Basin Management Plans base management decisions on the response of aquatic organisms to environmental stress. In contrast to the effects of degradation, the biotic response to restoration is less well known and poorly predictable. The timescale of the WFD (obtaining good ecological status in all surface waters by 2027) is over-ambitious. Future challenges include long-term monitoring of restoration measures to understand the

requirements for ecosystems to recover and prioritisation of measures according to re-colonisation potential.

Keywords: assessment, typology, uncertainty, monitoring, Heavily Modified Water Bodies, River Basin Management Plans, restoration, recovery

Introduction

The 1990s saw an emergence worldwide of holistic environmental management, integrated pollution control and countries embracing the ecosystem approach which combines natural and social sciences in tackling environmental problems (Apitz et al., 2006). This was most embodied in the Earth Summits in 1992 (Rio de Janeiro), 1995 (New York) and 2002 (Johannesburg) and the 1992 Convention of Biological Diversity. In these meetings, countries worldwide agreed to achieve environmental sustainability. Within Europe, this led to the proposal for a EU Directive on the Ecological Quality of Surface Waters which followed on from many countries adopting monitoring schemes and environmental quality objectives and standards. Since the 1970s, parts of Europe (e.g. UK and Sweden) had already shown a willingness to harmonise environmental measures to tackle trans-regional water quality issues (McLusky and Elliott, 2004). Following this, the regional seas agreements for the North-East Atlantic (the OSPAR Commission), the Baltic (the HELCOM commission) and the Mediterranean (the Barcelona Convention) were convened to achieve coordinated management of source catchments and receiving marine areas.

The European Directive proposal for the Ecological Quality of Surface Waters was never adopted, possibly because of its high ecological bias and inadequate consideration of socio-economic impacts. But this embryo of an idea eventually resulted in the drafting of the European Water Framework Directive which was finally adopted in 2000. The WFD had a precedent in the US Clean Water Act (CWA), published in 1972 and amended in 1977 and during the 1980s. There are clear parallels between the WFD and the CWA, in terms of objectives, implementation and ecological approaches. In both statutes, the status of water is important for a variety of uses and users, including bathing, outdoor recreation, industry and drinking (Hoornbeek, 2004). The policies arose from concerns about water status, where strong economic interests were often set against the diffuse interest of the general public. Policy solutions in this area generally included setting water quality standards, implementing discharge controls and minimizing the impacts of anthropogenic pressures on surface water quality (Hoornbeek, 2004).

The implementation of the WFD has been, and still is, a major challenge. Almost all EU Member States have spent considerable time and resources to develop tools, to gain the required data and to prepare River Basin Management Plans. In this context both the EU and its Member States have funded a large number of research projects, particularly in the areas of ecological assessment and catchment modelling.

The WFD has impacted various levels of environmental management of aquatic resources and has triggered the re-organization of water management by hydrological catchments, rather than by administrative borders, with the ultimate goal to improve the quality of surface water bodies. It has also been an important incentive towards harmonisation of classification and monitoring methods across Europe. The biotic communities of European surface waters are now the primary focus, used to assess the status of lakes, rivers and marine ecosystems and the success of management. The WFD has precipitated a fundamental change in management objectives from merely pollution control to ensuring ecosystem integrity as a whole. Deterioration and improvement of 'ecological quality' is defined by the response of the biota, rather than by changes in physical or chemical variables.

From a scientific perspective, the implementation of the WFD is greatly increasing knowledge on the ecology of European surface waters, particularly in regions which have rarely been investigated: approximately 1,900 papers have resulted from research projects associated with the implementation of the directive (query 'Water Framework Directive' in SCOPUS at 4/12/2009). Many methods to sample and investigate aquatic ecosystems have been developed and large amounts of data are being generated.

The underlying concept of the WFD and, in particular, the way it has been implemented in practice has received major criticism, from politicians, water managers and scientists (e.g. Moss, 2007, 2008; Dufour and Piegay, 2009). Here, we review experiences with the WFD implementation from the perspective of natural scientists involved in research projects and intercalibration working groups supporting the implementation process. We aim to provide a balanced review of both the successes and the problems encountered with implementation over the past 10 years and give recommendations on how to further improve the implementation process for the future. We particularly address three fields: (i) the development of assessment methods (including reference conditions, typologies and intercalibration); (ii) the design of monitoring programmes and how they are related to the assessment systems; and (iii) the consequences for river basin management plans (such as the implementation and success of restoration / rehabilitation measures).

Assessment of Ecological Status

The WFD was welcomed by many for its innovativeness and radical shift to measure quality of all surface waters using a range of biological communities rather than the more limited aspects of chemical quality (Moss, 2007). This was recognised as being a much more effective integrative way to measure ecological quality. This innovativeness did, however, come with a number of substantial challenges for ecologists in requiring complex and dynamic biological communities to be quantified into a single numeric score, rather than qualitative species lists, for reference conditions to be established from which to measure the degree of change, and for this all to be carried out within a large number of water body types. The uncertainty in the resulting quality classification and reference conditions also had to be quantified in a robust way. One major obstacle was the fact that no consistent biological datasets were generally available for lakes, rivers and coastal waters. A major achievement of the WFD is that many sampling and analysis procedures have been standardised across Europe (e.g. CEN, 2004), there has been investment in taxonomic training, and extensive monitoring programmes including physical, chemical and biological variables have been implemented. An overview of major implementation successes, problems and solutions is given in Table 1, while below we provide details on individual successes and obstacles.

Assessment systems: are we lost in complexity?

The requirements of the WFD concerning ecological assessment of aquatic ecosystems are both specific and general at the same time. Annexes II and V of the Directive contain many details, e.g. criteria for water body typologies and a range of specific components of five Biological Quality Elements (BQEs) and associated hydromorphological and physico-chemical elements to be monitored. While the WFD indicates what characteristics of the BQEs should be assessed (e.g. ‘abundance’, ‘community composition’) it does not specify which indices or metrics of these various elements should be used. The specification of metrics and indices for the different BQEs has been left to scientists in member states to propose, and this in turn has resulted in the age-old problem that those carrying out the monitoring are often unwilling to change from their usual practices. Most assessment systems existing in the year 2000 in the EU Member States were, however, not compliant with the WFD, as they were generally not reference-based (i.e assessed deviation from an acceptable baseline) or specific to water types.

Efforts to develop new methods fulfilling the complex requirements of the WFD were huge, and as the process was not organised centrally many national and international projects contributed (examples for lakes: Moss et al., 2003; Lyche-Solheim et al., 2008; rivers: Hering et al., 2004; Furse et al. 2006; Schmutz et al. 2007; coastal and transitional waters: Borja, 2005; Borja et al., 2004, 2007). No generally applicable European method for water body assessment resulted and methods developed differed between countries, between Biological Quality Elements and between water categories and types. Major differences existed in taxonomic resolution (species vs. higher taxonomic levels), the way of defining reference conditions, type vs. site specific assessment and the number and nature of indices (metrics) used.

A recent review of 252 WFD-compliant assessment systems published on www.wiser.eu/results/methods-db revealed that a large proportion (46%) of these systems target various forms of water pollution (acidification, eutrophication, heavy metals, pollution by organic compounds, pollution by organic matter). Other frequently addressed stress types are general degradation (19%), hydromorphological degradation (10%), habitat destruction (8%), riparian habitat alteration (5%), catchment land use (4%), flow modification (4%) and impact of alien species (4%), resulting in a higher diversity of stressors being assessed. Particularly for rivers, assessment metrics have often been selected based on their correlation to hydrological, morphological or land use parameters (e.g. Hering et al. 2004, Schmutz et al. 2007). In some cases assessment systems have been developed irrespective of stressors, comparing the present situation to historic data or least disturbed systems (e.g. Blomquist et al. 2007, Perus et al. 2007, Muxika et al. 2007).

Effects of different field and lab procedures, in many cases, are relatively minor (Furse et al., 2006, Borja et al., 2007) and in one case a common Europe-wide method has been developed (fish in rivers, Pont et al., 2006, 2007).

The unavoidable discrepancies in methodologies had to be managed by additional tools such as the intercalibration process. The developed assessment methods have often been criticised for being too complex, while much more simple parameters (such as water transparency) may give a sufficiently precise idea of the ecological status (Moss et al., 2003; Peeters et al., 2009). Yet this criticism does not offer alternatives that are compliant with the WFD legislation. Peeters et al. (2009) provided convincing arguments that transparency suffices for determining the eutrophication status of lakes, although they only illustrate their case on a restricted set of water-bodies – very shallow, lowland lakes. No evidence is given that the approach is applicable to other lake types or lakes where eutrophication may not be the key pressure. The strength of the WFD approach (monitoring a range of biotic communities) is that it potentially addresses complex mixtures of stressors in very different regions and water-body types.

Advocates for simplicity in the assessment systems also argue that the breadth of current approaches developed do not encapsulate the concept of a healthy functioning ecosystem. The requirements of the WFD assessment schemes outlined in Annex II and V predominantly relate to structural elements rather than functional ones. Consequently, many of the new metrics developed focus on taxonomic indices, rather than ecosystem function (e.g. de Jonge et al., 2006). Although it could be argued that taxonomic metrics are fundamentally an expression of function, future research could explore further how structural elements could be used more explicitly to represent system functioning (e.g. macrophyte growing depth as an indicator of benthic vs. planktonic production, ratios of invertebrate functional feeding groups). Moss (2008) argues that key features such as nutrient parsimony, connectivity and resilience to change should be included. There are certainly different ways of assessing ecosystem health but as the annexes of the WFD are explicit concerning biotic data to be included into assessment systems taxonomic indices of adequate confidence and precision can not be avoided, irrespective of the potential worth of alternative approaches.

A major achievement of the WFD has been the development process itself. In all Member States experts working on different organism groups and ecosystem types considered 'the best approach' for monitoring and developing ecological classifications. The large number and variety of people involved in the development of assessment systems for the WFD can be seen in a recently generated overview of European assessment methodologies on www.wiser.eu/results/methods-db.

It is hard to argue against the fact that biomonitoring methods and data quality have improved overall. The fact that different assessment systems evolved across Europe reflects the diversity of water body types and pressures: in some countries and ecosystem types single stressors which are easy to assess predominate (e.g. organic pollution or eutrophication), while in other cases a complex mixture of stressors affect water bodies (e.g. nutrient enrichment, hydromorphological degradation, toxic substances, overfishing). Ecological knowledge of different organism groups varies across Europe. In Northern Europe most aquatic species and their ecological preferences are known, while the aquatic stages of many species occurring in Southern European waters are still not described (Schmidt-Kloiber et al., 2006).

In conclusion, technical implementation of the WFD Annexes is a complex process, but the use of several quality elements and establishment of typologies and reference conditions is a major improvement. The resultant schemes are probably more complicated than what the authors of the WFD intended. The effort required for developing assessment methods was, however, grossly underestimated and, therefore, assessment methods were often not available before River Basin Management Plans had to be drafted in 2008-2009. On the other hand, the development process and the resulting methods have led to a new understanding of applied aquatic ecology in Europe; knowledge that is now not restricted to a small group of researchers. Indeed, technicians, water managers and, to some degree, stakeholders and politicians, have contributed to the process and learned to communicate despite educational and cultural differences. So, maybe the greatest value emerged from the process itself.

Uncertainty in assessment

A central element in WFD-compliant assessment systems is the estimation of uncertainty. This builds on the understanding that there is no definitive means in bioassessment and that all results are influenced by several sources of variability and errors, for example variability in sampling and laboratory analysis, seasonal and geographical variability (Clarke and Hering, 2006; Carstensen, 2007). For this reason, ecological status classification results should always be given in terms of probabilities. Today only a small proportion of assessment systems have put this into practice. Including uncertainty estimation into assessment schemes is a major challenge of the next phase of WFD implementation. The underlying statistical principles are relatively simple and appropriate tools for uncertainty estimation are available (e.g. Clarke and Hering, 2006; Carstensen, 2007) but data are needed which address the individual sources of error, such as differences between investigators and sampling equipment/analysis, as well as temporal (diurnal, weather event-related, seasonal) and spatial (representative sampling location) variation of sampling, affecting the distribution of the assessment results. These principles apply to all assessment systems, even to methods, which are very simple to apply such as those suggested by Moss (2008). For example, the WFD has been a major driver in improving our understanding of the effect of sampling frequency and location on annual estimates of total phosphorus and phytoplankton chlorophyll a (Carvalho et al., 2006; 2007). Given quantitative information of these sources of uncertainty, the likelihood of different status classifications can be computed. More challenging, however, is to convey the concept and principles of uncertainty to water managers: that it is more appropriate to know the amount of error affecting an assessment method than to give results with an unknown or unrealistic precision. If the major sources of error are known, they can potentially be minimised through the re-design of sampling schemes (additional sampling sites or frequency), through improved training by operating procedures, CEN (European Committee for Standardization) guidance, taxonomic training or through the use of model-based assessment methods (Pont et al., 2009). Though there is no central overview available, taxonomic training has been implemented in several countries in connection with the WFD: In Germany, the German Limnological Association has offered 35 training courses on different organism groups (http://www.dgl-ev.de/arbeitskreise/ak.taxonomie.html), additional courses in Germany have been offered by the Senckenberg Institute. In Austria training courses cover all BQEs (http://wasser.lebensministerium.at/article/archive/5659). In Finland, training on phytoplankton taxonomy has been carried out by the Finnish Environment

Institute in collaboration with the Finnish Phytoplankton Society. Also regular intercalibrations of phytoplankton analysis have been conducted. The Quality Assurance of the phytoplankton counting has been ensured by reference laboratory activities as described by Lepistö et al. (2009). Marine biologists have agreed on common taxonomical standards (http://www.marbef.org/data/erms.php) which is now the basis for identification by most labs. Inherent in the discussions of uncertainty is the realisation that scientists will have their methods and approaches subjected to legal and political scrutiny. The determination of ecological status, and thus the need to invest large amounts of money to remediate problems, is influenced by the uncertainty in defining status, especially when metric results are close to the good/moderate class boundary. Thus any Member State that is taken to the European Court through infringement procedures related to doubtful assessment methods would have to demonstrate the robustness of its methods. Furthermore, there is concern about the capacity within monitoring agencies across Europe to design and implement monitoring programmes with sufficient sampling to provide a proper basis for uncertainty estimation. This concern is re-enforced by the change of many national Environmental Protection Agencies over the past decades from executive bodies of aquatic monitoring to merely administrative bodies with quite a remote sense of the need for scientific rigor in the ecological status assessments.

Typology: is it needed?

According to the WFD, ecological assessment has to be 'type specific', i.e. water bodies should be grouped according to their physical and morphological attributes, such as salinity, alkalinity, catchment size or altitude/depth. With the experiences gained during the WFD implementation process it is clear that the use of water body types is a simple and appropriate tool for water managers and the general public to better understand the natural differences in aquatic communities and consequently differences in restoration targets. On the other hand, typologies are coarse delimitations of naturally continuous gradients across a wide range of ecosystem characteristics. In reality many environmental parameters influence community composition, even when human-induced stress is not considered (Sandin and Verdonschot, 2006; Aroviita et al., 2009). The WFD allows any natural environmental parameter influencing communities to be included in the typology system (System B, Annex II), but there is always a trade-off between having all environmental factors included and having a manageable typology system. There is no compilation of the typologies used by the European member states available but most likely the individual typologies are not comparable at all.

One way forward is a relatively simple approach consisting of broadly defined types (e.g. Moss et al., 2003 for lakes), which coarsely discriminate ‘common types’ to be used in the intercalibration process. Such types have been defined for lakes, rivers and coastal waters, but still need to be determined for transitional waters (Borja et al., 2009a). The alternative is a sophisticated typology reflecting relatively minor natural ecological gradients and thus fine-scale differences in community structure as described by Verdonschot (1995) for rivers in the Netherlands, Lorenz et al. (2004) for rivers in Germany and Hull et al. (2004) for coastal and transitional waters in the UK. Site-specific assessment (prediction systems) might be the ideal solution, as this approach incorporates the individual characteristics of a site, rather than adopting a standard set of descriptors partitioning natural variability. Recent studies suggest that site-specific assessments have higher sensitivity, particularly for water bodies close to typology boundaries and in the absence of undisturbed sites for a water body type (Clarke et al., 2003; Pont et al. 2006; Cardoso et al., 2007; Aroviita et al., 2009; Carvalho et al., 2009).

In conclusion, it is emphasised that parameters relevant for typology are among the major sources of uncertainty in ecological assessment. The more specific assessment systems are better if they have been corrected for typological differences. While for the coarse evaluation of ecological status, and communication of results to managers and the public, broadly defined types might be sufficient, the logical endpoint for a sophisticated assessment method will be site-specific prediction systems, although not strictly WFD-compliant.

Intercalibration: Comparing the incomparable?

The authors of the WFD had in mind a simple assessment system. Likely they had the vision of just a few assessment metrics to be applied across Europe – this proved not to be realistic nor achievable: stressors affecting aquatic ecosystems differ between regions, and the effects of different stressors (e.g. acidification and eutrophication) could not be assessed with the same metrics. Water body types not only differ in terms of size and catchment geology, but also in their species pools and the bioindicator taxa present. Unavoidably, sampling methods also differ between types, e.g. small and large rivers. Between regions, knowledge on the taxa occurring differs greatly (Schmidt-Kloiber et al., 2006). Therefore, uniform taxonomically-based assessment methods could not account for all these differences to be applicable throughout Europe. Alternatively, ecological assessment could have been based on simple parameters, such as water transparency and catchment land use (Moss et al., 2003; Peeters et al., 2009).

One of the most important obstacles for implementing a harmonised assessment is that biomonitoring traditions differ between countries (especially for invertebrates). Countries having well established biomonitoring systems were resistant to change, in particular those countries having long time series. These differences have led to several methods reflecting both a variety of European water bodies and biomonitoring history. The logical consequence was that methods used for the WFD have to be intercalibrated, a comparison process which was already planned for in the WFD (Annex V, section 1.4.1).

The first intercalibration was a pilot exercise with an unknown outcome and had to compare many methods, many of which had not been fully developed (Heiskanen et al., 2004), although some experience in comparing a limited number of assessment methods using correlation methods existed (e.g. Ghetti and Bonazzi, 1977; Friedrich et al., 1995; Stubauer and Moog, 2000; Krause-Jensen et al., 2009). The WFD intercalibration approach was originally thought to be based on comparison of member states' assessment methods on a small number of sites; however for statistical reasons this was not useful. Therefore, other options were developed (Common Implementation Strategy, 2005), in which the compilation of a dataset of sites covering the whole pressure gradient was recommended. One of these options ('Option 2') is based on 'common metrics', against which national methods are compared.

For some BQEs and water categories, such as benthic invertebrates in coastal waters (Borja et al., 2007, 2009a) and phytoplankton biomass in lakes (as chlorophyll a) (Poikane, 2009), the intercalibration results were surprisingly clear: most of the assessment systems give the same pattern. For other BQEs, such as phytoplankton composition in lakes, the first intercalibration results differed so much for certain regions (Central-Baltic GIG) that the results were rejected by the Commission from the Intercalibration Official Decision. This was largely a result of the diverse array of metrics produced across Member States. For some BQEs, such as fish, and one water category (transitional waters) the assessment systems had not been sufficiently developed to allow any intercalibration results in the first phase (2004-2008).

The first phase of the intercalibration exercise has been subject to two separate scientific reviews on coastal / transitional waters and on lakes / rivers, which generally agreed with the finally selected approaches, e.g. the use of common metrics and the use of bands of acceptable boundary values. However, several critical points were raised, in particular it needs to be ensured that reference conditions are set in a harmonized way, intercalibration is done separately for different stressors, and inter-annual variability needs to be taken into account. Due to these shortcomings the EC extended the intercalibration process with a second phase (2009-2012) to allow completion of intercalibration for all BQEs in all water categories. A new intercalibration guidance and new annexes have been drafted, addressing more harmonised procedures to set reference conditions and class boundaries and to compare the outcome of individual intercalibration exercises.

For this second phase of the intercalibration exercise three main problems remain: (i) there is still a significant delay in the process, which is due to the slow development of assessment systems in many countries; (ii) the number of individual intercalibration exercises is very high (number of GIGs * number of BQEs * number of water categories leading to > 100 exercises); and (iii) dissemination of intercalibration results is difficult. Although the intercalibration methods used are basically simple the process itself has been composed of several steps and is relatively complex. Combined, these problems have often led to the fear among water managers that intercalibration will have significant impact on already finalised steps of WFD implementation used as a basis for the first River Basin Management Plans, e.g. on the identification of which water bodies actually need to be restored and the associated planning and reporting requirements.

Merging assessment results: The funnel effect

Summarizing all sources of variability into an ecological assessment of a water body results in two types of errors: type I errors (detecting a difference when no real difference exists) and type II errors (not detecting a difference which is real). As type I error increases when type II error is reduced and vice versa, provided the number of observations remains unchanged, both of these errors cannot be eliminated unless the entire population is sampled. They are best managed by giving probabilities, i.e. the likelihood of a site to fall into a status class (Clarke et al., 2003).

One of the challenges of the WFD results from the combination rules stipulated. In general, different organism groups are sampled per water body and assessed independently. The lowest score of all assessment results determines the overall ecological quality class (i.e. the assessment defaults to the lowest category, the ‘one-out, all-out’ principle; see WFD Annex V, section 1.4.2 (i) and WG ECOSTAT 2003).

This procedure is prone to reduce type II errors (i.e. reducing the likelihood that a water body is classified as good status, when in reality it is below good status). The ‘one-out, all-out’ principle is thus in line with the precautionary principle, and will provide sufficient protection for the most vulnerable BQE to the most dominant pressures. At the same time this principle will also tend to inflate type I errors (concluding that a water body is below good status, even if the water body in reality has good status), thus posing a risk of implementing measures where they are not strictly needed. For instance, if three BQEs in a good-status water body are sampled and one of these results is affected by a type I error (e.g. wrongly classified as moderate status), the final result (moderate status) will be determined by the error – irrespective of the fact that the two other results are correct (good status). As a result, the ‘one-out, all-out’ principle increases the likelihood of deriving a lower status class by sheer randomness, whereas the risk of misclassifying to a higher status than the actual state becomes less likely (Sandin, 2005). An example from Germany is given in Table 2, showing that a much larger proportion of sites fail the good status objective when the one-out-all-out rule is used compared with when only one BQE is used.

The ‘one-out, all-out’ principle has been criticised by several authors (Borja and Heinrich, 2005; Sandin, 2005; Sondergaard et al., 2005; Borja et al., 2009c; Tueros et al., 2009) for these statistical reasons. Furthermore, it contrasts with the ecosystem approach the WFD is pursuing, as it is scientifically difficult to justify that a single component determines the quality of an ecosystem. As the legislation is clear in terms of the ‘one-out, all-out’ principle there is no simple way to avoid this problem. Options to reduce type I errors include: (i) the choice of confidence levels for the different BQEs in a way to minimise the risk of type I errors (Carstensen, 2007); (ii) increase of sampling frequency or density to reduce the variation in each BQE; (iii) omitting BQEs with too high variability from the assessment (the latter is also recommended by the WFD). Future amendments of the WFD may consider alternative combination rules (see Borja et al., 2004, 2008a, 2009b) and should require estimates for the degree of type I and type II errors.

Assessment of heavily modified water bodies (HMWB)

The WFD requires Member States to distinguish between ‘natural’ and ‘heavily modified water bodies’ (HMWBs). The latter are designated as having an acceptably lower ecological status as the result of hydromorphological pressures, which cannot be removed because of the high social or economic cost. Because of this, the quality targets for HMWBs are ‘good chemical status’ (compliant to natural water bodies) and ‘good ecological potential’, pragmatically defined as the ecological quality expected under the conditions of the implementation of all possible measures (see Borja and Elliott, 2007). This may result in significantly reduced ecological quality targets. The designation process of HMWBs is composed of several steps and involves a certain level of complexity (Common Implementation Strategy for the Water Framework Directive, 2002). Nevertheless, a significant proportion of European water bodies has been designated as HMWB due to hydromorphological degradation; in four member states (Netherlands, Belgium, Slovak Republic, Czech Republic) more than 50% of the water bodies were designated as HMWB. With the exception of these first four, member states have on average provisionally identified around 16% of their water bodies as heavily modified and artificial (Commission of the European Communities, 2007).

Two different approaches towards ecological assessment exist for HMWBs: the Prague approach (Kampa and Kranz, 2005) which is mainly based on measures and the Common Implementation Strategy guidance approach more strongly involving biological assessment (CIS Working Group 2.2 on Heavily Modified Water Bodies, 2003). As HMWBs are not exceptional cases the comparability with assessment results to those obtained for natural water bodies should be guaranteed. From our point of view, the assessment of HMWBs should therefore be based on the same metrics as for natural water bodies. The quality targets should be adapted on a case-by-case basis, in some cases removing those BQEs which are directly affected by hydromorphological pressures (e.g. macroalgae and angiosperms in transitional waters modified as harbours, which lack suitable habitats after massive dredging), while keeping those that are most sensitive to the other pressures acting on the HMWBs.

Monitoring systems

The assessment systems discussed above are the principal tools for monitoring ecological status under the WFD, which have now been implemented in all EU member states. The WFD distinguishes among three types of monitoring (see Borja et al., 2008b): (i) surveillance monitoring, to assess long-term changes resulting from widespread anthropogenic activity; (ii) operational monitoring, in order to establish the status of those water bodies identified as being at risk of failing to meet their environmental objectives; and assess any changes in the status of such water bodies resulting from the programmes of measures; and (iii) investigative monitoring, carried out where the reason of any exceedance for ecological and chemical status is unknown; where surveillance monitoring indicates that the objectives for a water body are not likely to be achieved (and determine the causes); or to ascertain the magnitude and impacts of 'accidental' pollution.

The implementation of the monitoring programmes is a great achievement, as for the first time comparable pan-European data sets to assess ecological status of surface waters are being obtained as a fundamental basis for restoration of impacted aquatic ecosystems (Ferreira et al., 2007). In addition to the development of assessment systems, the establishment of harmonised monitoring programmes is still a challenge, since the design of monitoring programmes reported to the Commission is highly variable in terms of station density, sampling frequency and choice of BQEs. From our point of view the following issues should be regarded to further strengthen the programmes.

The data: Big deal or big mess?

One of the major consequences of the WFD is the acquisition of large amounts of biological information on the status of European surface waters, information that may improve our knowledge of the structure of the communities inhabiting these ecosystems. Potentially, these data could contribute significantly to other objectives in addition to those of the WFD, e.g. for monitoring the effects of emerging stressors, for improving our knowledge of species distributions and species invasions, for understanding broad scale drivers shaping community assemblages, for Habitats Directive/Natura 2000 species inventories and biodiversity records. However, as with the variability of methods employed for collecting data, the data structure, quality and quantity are quite variable. This applies to the underlying taxonomy and taxonomic identification codes, taxonomic resolution, density of sampling sites, sampling frequency and data storage. As an example, according to Commission of the European Communities (2009) there are 428 river surveillance and operational monitoring sites in Hungary (corresponding to a density of 4.6 sites/1,000 km^2), but 2,731 sites in Ireland (38.9 sites / 1,000 km^2). The density in Poland is 9.0 sites/1,000 km^2, but 49.0 sites/1,000 km^2 in the UK. While all these data will be useful to guide regional restoration programmes, Europe-wide comparisons can often be made on the coarsest resolution. There are some exceptions to this, as part of the EC REBECCA Project, chemical and biological data from more than 5000 lakes in 20 European countries were compiled into pan-European databases incorporating data from phytoplankton, macrophytes, macroinvertebrates and fish (Moe et al., 2008).

At present, Europe-wide comparisons are furthermore limited to data on the overall ecological status and selected metrics, as the original data (e.g. taxa lists) are not being stored centrally, which limits their potential for large-scale analyses and for purposes beyond the WFD. There are, however, promising steps. WISE (Water Information System for Europe; http://water.europa.eu) produces Europe-wide maps of water quality, currently only based on environmental variables. The European Environment Agency (EEA) is now also considering producing ecology-based WISE maps, and their test data request in 2009 resulted in more than 34,000 data records on individual BQEs from almost 10,000 sites in 17 countries. Moreover, the European Commission and the EEA have launched the web-based SEIS (Shared Environmental Information System), which will simplify the reporting and accessing of environmental information. A useful future step would be to link data from all member states and from research projects to these systems without transferring data to any central database. This would be a major exercise, however, it would be worthwhile to make maximum use of the huge investment in biological recording.

Monitoring: What is required by the WFD and what is useful?

Most countries focussed on operational monitoring: according to the Commission of the European Communities (2009) the number of operational monitoring sites is higher than the number of surveillance monitoring sites in 17 out of 25 reported EU member states. Therefore, the WFD approach is clearly orientated towards restoration: the monitoring results should reveal if and what type of restoration is needed and, in the future, if restoration was successful. The shortcoming of the operational monitoring is that it does not reveal long-term trends, which are independent of the local situation. Over-arching trends, such as the impact of emerging stressors (climate change, land use change, new pollutants), changes in species distributions and ecological processes would be better revealed by a network of reference sites.

There are, however, exceptions to this at the national level. In France, the total number of river monitoring sites in 2000 was 1,560 and has been relatively constant since 1987. Most sites were located in the downstream part of rivers and water agencies focused mainly on chemical status. In 2007, the total number of monitored sites was 2,860: 1,276 for surveillance monitoring, 790 for operational monitoring and 794 for both monitoring programmes (OIWater 2009). This total number reached 4,337 in 2008, mainly in relation to an increase in operational monitoring effort. Within the surveillance monitoring network, the site density per kilometre of river is now comparable between downstream and upstream reaches, and the ecological status is assessed using 895 variables: water chemistry, biological elements and hydro-morphological characteristics. To assess any long-term changes in reference conditions in relation to large scale environmental change (e.g. global warming), about 400 sites characterized by a low level of human pressure and good biological quality have been selected to create a permanent reference condition monitoring network.

The EEA EIONET or WISE stations may provide such a network Europe-wide, since these are now being based on the WFD surveillance monitoring stations of the Member States. This ‘central monitoring network’ should address both high status sites to analyse long-term trends, irrespective of regional peculiarities, and a well-designed subset of degraded and restored sites to monitor the effects of both degradation and restoration over time. Ideally it should also be linked to the network of Long-Term Ecosystem Research sites (LTER; http://www.lter-europe.net).

The WFD and other European legislation

The WFD aims to link with some pre-existing EU directives and replace others. There are several other directives which also aim to determine whether or not an area is affected by human activities. For example the Marine Strategy Framework Directive (MSFD), the Urban Wastewater Treatment Directive (UWWTD), the Nitrates Directive (NiD) and the Habitats and Species Directives (HSD) all require member states to check if an area is adversely affected by pressures, with the ultimate goals to remedy any problems. The objectives of these directives are not consistent in terms of terminology – for example, the WFD, the HSD and the MSFD expect areas to fulfil ‘good ecological status’, ‘favourable conservation status’ and ‘good environmental status’, respectively (Mee et al., 2008). For the directives to be harmonised, there is a presumption that these status classes are equivalent, especially as the designated areas can overlap, including also the sensitive areas and the vulnerable zones of the UWWTD and NiD (see Common Implementation Strategy for the Water Framework Directive, 2009). However, some areas are now being designated as being HMWB and yet being in favourable conservation status (e.g. the upper part of the Humber Estuary, eastern England). Accompanying this is a debate regarding the geographical limits of the directives, in particular where the WFD stops at sea and where the MSFD starts. As yet, these anomalies need guidance before scientists are asked to determine whether 'good environmental status' and 'good ecological status' (and favourable conservation status) are synonymous.

Table 3 shows how different directives, conventions and thematic strategies are related. Hence, the new MSFD (Commission of the European Communities, 2008; Mee et al., 2008), as well as the WFD, constitutes an umbrella over the remainder of actions and directives, at the European and eco-regions level. Most of the existing directives are related to the lowest level of the ecological organisation (species, habitats). However, WFD and MSFD are more complete in terms of ecological structure, environmental quality and more integrative in terms of ecological assessment (Borja et al., 2008a).

Both directives integrate biological factors with physiographic, geographic and climatic factors and physico-chemical conditions resulting from human activities. While the WFD focuses on ecological status, measured by the structure of each of the BQEs and supporting elements, the MSFD takes into account structure, function and processes in marine ecosystems. Hence, the MSFD is potentially a more integrated approach to the management of European seas, resources and ecosystems, promoting conservation and sustainable use of marine systems (Borja et al., 2008a).

River Basin Management Plans

Despite the potential value of the WFD monitoring data for many other purposes ranging from biodiversity analyses in support of the Habitats Directive to basic ecological research, the principal aims are to identify restoration needs and to guide restoration measures. The instruments to implement these measures are River Basin Management Plans (RBMPs). In the framework of River Basin Management Plans the costs for monitoring will be negligible relative to the costs of restoration measures. Operational monitoring should, therefore, be regarded as an integral part of a RBMP. The linkage between monitoring data and the designation of measures has not yet been fully explored but initial studies allow us to outline the following recommendations.

Ecological assessment and River Basin Management Plans: The challenge of bridging ecology and management

One of the most innovative aspects of the WFD is to base management decisions on the ecological effects of pollution (or other stressors) rather than the pollution itself, acknowledging that sensitivity and resilience to pollution varies substantially across ecosystems. The associated challenge is to translate data on biotic communities into information for restoration measures. This has now, in principle, already been done for the first RBMPs. In reality, however, the links between ecological status and restoration measures are obscure in many plans, due to the delayed development of assessment systems and initiation of monitoring programmes. Moreover, there has been no central guidance available on how to transfer ecological assessment results into management decisions.

In many countries there was an intense consultation process in the drafting phase of the River Basin Management Plans. Positive examples of a transparent consultation process are Northrhine-Westphalia (a federal state in Germany, see http://www.flussgebiete.nrw.de/Mitwirkung/index.jsp) where round-table discussions in the individual river basin districts were organised involving a wide variety of stakeholders and the Basque country in Spain were similar exercises have been performed over a three-year period (http://www.uragentzia.euskadi.net/u81-0003/es/contenidos/informe_estudio/planificacion_dma/es_doc/indice.html). In Finland the stakeholder's involvement has been organised by regional environmental centres that have established cooperation councils. A critical study of the participatory process was made by NGOs (Laurinolli 2007). In general they found that stakeholders were well represented in the process. However, during the first consultations the NGOs, the general public as well as the media had not properly engaged in the process, possibly because they had not properly understood the importance of the planning process for water management in the future. The Swedish RBMPs demonstrate extensive and transparent involvement of local, regional, national and international stakeholders, including NGOs. Here, universities have been involved in the training of local and regional water managers, the meetings held and the comments given are publicly available and summarised, accounts are given on how the comments have been taken into account when revising the RBMPs and conclusions on the lessons learnt are presented. Most river basin districts have established permanent organisational structures called water councils for the large majority of separate river basins within the RBDs. These water councils are comprised of representatives of a series of organisations (environmental NGOs, local farmers, local enterprises, citizens) and have given comments on the various parts of the local RBMPs.

Linking ecological data and restoration measures is rather straightforward when dose-response relationships are simple and well-known, e.g. for organic pollution of rivers. It is difficult, however, in case of stressors, whose effects are less well known, and especially in the case of complex multiple stressor situations.

As water quality has been improved in many parts of Europe (Lyche-Solheim et al., 2010), river rehabilitation nowadays focuses more on restoring habitats, and it is widely expected that benthic invertebrates, macrophytes and fish will respond positively. However, most restoration measures have targeted relatively short river stretches and consequently biological recovery has not been achieved. This lack of restoration success is probably due to the need for more widespread improvement of habitat quality on the catchment scale and also on recolonization potential (Jähnig et al., 2009, Palmer et al., in press). In the case of transitional and coastal waters, the ecological assessment exemplifies the problem of transboundary pollution pressures and the wider effect of stressors. Thus, transitional waters receive pollution from the whole catchment and may thus act as both a source to the sea and a sink from the catchment, especially as they may be low energy, depositing areas and therefore effects are exacerbated. In contrast, the quality of coastal waters is not only affected by river catchments but also by stressors in other marine areas. Hence, in order to design an appropriate programme of measures, water managers are charged with untangling these various pressures on a given area, and, therefore, will need significant scientific support.

For the first cycle of River Basin Management Plans, biological assessment results were often not available prior to the planning process. Therefore, ecological assessment and planning were partly disentangled. An overview of all River Basin Management Plans can be found on http://cdr.eionet.europa.eu/ and on http://ec.europa.eu/environment/water/participation/map_mc/map.htm, covering the entire range from very general formulations of environmental targets to precise planning of restoration measures based on the results of the monitoring programmes. Positive examples where management decisions have been based on large-scale considerations of the ecological status and the requirements of the Biological Quality Elements are the German federal states Schleswig Holstein (Brunke and Lietz in press) and Thuringia (Arle and Wagner, in press) and the Dutch method to derive the Good Ecological Potential in Heavily Modified Water Bodies (e.g. Lammens et al. 2008). General suggestions which measures affect which organism group are amongst others found in Kail and Wolters (in press). A promising example from marine ecosystems can be found on http://www.uragentzia.euskadi.net/u81-0003/es/contenidos/informe_estudio/diagnostico_agua/es_doc/indice.html.

To make the maximum use of the biological data presently being recorded it is essential to make dose-response relationships between stressors and the biotic response available to all river basin managers well before the design of the second cycle of RBMPs and provide scientific guidance on the most simple and effective restoration measures appropriate to enhance ecological quality.

There is a danger that some of the measures listed in the RBMPs cannot be implemented in practice due to a lack of political instruments to enforce their implementation, e.g. to seriously reduce diffuse pollution sources. Only the coming years will show which measures are actually implemented, and which political instruments need to be developed that will guarantee their enforcement.

Is good status enough?

The aim of the WFD is to reach good status for all water bodies which are not designated as 'heavily modified'. Good status is defined as a 'slight deviation from reference conditions' and moderate status is 'moderate deviation from reference conditions'. Hence scientists are charged with determining reference conditions in quantitative terms, as well as the meaning of 'slight' and 'moderate'. The first intercalibration revealed that for some BQEs and water categories there is a common understanding amongst scientific experts of the meaning of 'good status' – despite large differences in assessment systems.

The question arises what will be gained if 'good status' of the majority of European water bodies will be achieved? Water bodies in good status will have an acceptable water quality and will be characterised by the absence of other severe stresses. But, are they sufficient to maintain European aquatic biodiversity and associated functions and services?

In terms of protecting aquatic biodiversity high status sites may play a key role: Species richness and the number of sensitive species differ greatly between 'good' and 'high' status sites. For example, Aroviita et al. (2009) noted clear differences between high and good quality classes, with fewer occurrences and lower abundances of threatened species at sites classified as good compared to high ecological quality. Individual high status sites are not necessarily characterised by a high alpha-diversity (e.g. in case of ultra-oligotrophic lakes and marine water bodies), but there are several species and possibly genotypes restricted to sites of high ecological quality. High status sites, therefore, are required to maintain a high level of beta- and gamma diversity. The resulting need for the protection of high status sites is somewhat implied by the WFD which prohibits the deterioration of ecological status.

A possible solution would be a network of 'high status sites' as key areas for protecting aquatic biodiversity. These could also serve to underpin how natural (climate) variability affects the uncertainty in our assessment of type I and II errors of putative perturbed sites.

<u>How does ecological status respond to restoration?</u>

WFD monitoring for the first River Basin Management Plan was focussed on assessing the present status of a water body. The ultimate aim of monitoring, however, is to detect change, i.e. the deterioration of ecological status or the improvement following restoration / rehabilitation. Assessment systems should therefore give general guidance on the measures required.

The challenge is to predict how the biota will respond to restoration and what management actions are best suited. These questions are easier to answer for lakes and marine ecosystems, which are predominantly affected by eutrophication and where the main restoration measure is the reduction of nutrient load. It is more difficult for rivers, which are also affected by hydromorphological degradation on different spatial scales and transitional waters where increased turbidity and a naturally poor light regime complicates the response. The concepts of how organism groups respond to restoration measures are clear (rivers: Hering et al. 2006; lakes: Jeppesen et al. 2005; estuaries and marine areas: Elliott et al. 2007). However, there is a lack of empirical data on relevant geographical and long-term scales required for assessing restoration / rehabilitation success. It is unlikely that operational monitoring can be used to obtain this type of knowledge as sampling frequency and locations are often too coarse; usually there is a single sampling site per water body, which may cover several kilometres of river length.

One possible solution would be dedicated monitoring of a subset of water bodies subject to restoration measures with more sampling sites and higher sampling frequency both before and after restoration. Ideally, restoration studies, and indeed all studies of disturbance and recovery, should be based on deviation from an undisturbed condition. A robust statistical design would include three types of sites: (i) restored sites, (ii) target or control (reference) sites, and (iii) sites similarly impaired as those restored but not restored (e.g. Downes et al., 2002). Experiences with the effects of restoration should be collected centrally (ideally Europe wide) and be made available for all users.

Ecological and political timescales

The aims of the WFD are ambitious and clearly defined: By 2015, all water bodies (with the exception of heavily modified water bodies) need to reach good status, with a possible extension for another 12 years. There is, however, overwhelming evidence that across much of Europe even this extended time frame may not be sufficient to reach ‘good ecological status’. Recovery of biotic communities requires the implementation of measures and the response of the ecosystem – both steps need many years, sometimes decades. Jones and Schmitz (2009) give a broad overview of time scales required for recovery. The authors reviewed 240 recovery studies across terrestrial and aquatic ecosystems and found mean recovery times of 10 to 20 years for freshwater, brackish and marine systems. In all systems, macrophyte recovery was slowest, except for rivers where functional recovery required most time. But the authors also stressed that pre-perturbation data were available for only 20% of the reviewed studies, a factor that rendered the assessment of recovery in 80% of the studies rather subjective.

Restoration measures in rivers mainly depend on the availability of floodplain area. It is a long process to acquire space for the river floodplain. State-of-the-art ‘passive’ restoration requires the development of near-natural vegetation in the floodplain, which may take several decades (Kail and Hering, 2005). Reducing eutrophication in all water categories may require changes in land use on large scales. As a consequence, water and habitat quality required for good status can not be achieved everywhere within one or two decades.

According to Jeppesen et al. (2005) reduced external phosphorus loading in lakes resulted in a new equilibrium for total phosphorus within 10 to 15 years, restoration of many biological variables generally took much longer. For four well-studied coastal ecosystems, Duarte et al. (2009) did not observe a return of simple biological variables (such as chlorophyll concentration) following the reduction of nutrient loads over a time span of two decades. In some marine ecosystems nutrient residence times are on the order of decades, like in the Baltic Sea and, therefore, significant effects are unlikely to be achieved for the whole marine area by 2015. However coastal bays, lagoons and archipelogo areas that have lower residence times and are generally impacted by land-based nutrient inputs; here, effects of River Basin Management Plans are potentially visible within the WFD implementation time scale of 5 to 15 years (Kauppila et al., 2005). There are several examples, both in coastal and transitional waters, in which recovery can take between 2 and 15 years after a pressure is removed (Borja et al. 2006, 2009b, 2009d; Uriarte and Borja, 2009). Perhaps the best example of restoration in transitional waters is the recovery of the fish community in the Thames estuary passing through London. It took several decades to acquire a full species complement after starting from a state without any fish in the 1960s (McLusky and Elliott, 2004).

Sensitive species, which are required for a ‘good ecological status’, have been brought to extinction in entire catchments, particularly in densely populated areas throughout Europe. Restoring water quality and habitats does not automatically mean that sensitive species will reappear. It depends on source populations, colonization paths – and sufficient time acknowledging that we have been degrading aquatic systems in Europe since the start of the industrial revolution in the early 1800s.

In conclusion, we cannot expect European aquatic ecosystems to fully recover within 15 or even 30 years from over a century of degradation. Where restoration measures and land use changes can be implemented rapidly there will in many cases be improvements of ecological status within this time span, although not necessarily all the way to good status. The overall aim to reach good status for most European water bodies is ambitious but not realistic in the given timeframe.

How do we deal with emerging stressors?

The WFD and corresponding assessment schemes mainly focus (and were designed to focus) on ‘traditional stressors’, such as eutrophication, organic pollution, acidification, toxic stressors and to a lesser degree hydromorphological pressure. Other stressors have more recently come into focus, such as such as climate change, siltation, new toxic substances and alien species. Diagnostic metrics are currently only available for common types of degradation. Therefore, there is a need to focus on whole ecosystem and community structure and functioning. Pollution response science assumes that changes to individual organisms due to pollution will be transmitted through the ecosystem and manifested at the community level. However, we know that systems have an inherent ability to absorb stress (Elliott and Quintino, 2007) and so effects of stressors on individuals may not necessarily be reflected in the metrics currently being used for the WFD. The science now needs to be developed to look at response trajectories and the resilience of ecosystems (Elliott et al., 2007).

One possible solution to include climate change effects is to assess the impact of climate change on existing WFD metrics and then adjust the existing assessment systems accordingly. Another way is to add ‘climate specific components’ to assessment systems, e.g. metrics particularly reflecting the temperature sensitivity of species. More generally, assessment schemes should allow for a certain degree of flexibility, to address changes which will be relevant in the future. The overall design of WFD compliant assessment is well suited to detect the effects of emerging stressors, as changes in biotic communities irrespective of their causes are monitored.

Conclusions

The EU Water Framework Directive is a very ambitious piece of environmental legislation which places aquatic ecology in the centre of water management. The performance of ecological assessment under the WFD varies between regional, national and European scales, across seasons and ecosystems types (lakes, rivers and coastal/transitional waters).

The monitoring data can directly support RBMPs on a regional scale. These data will provide guidance for restoration measures and evaluate their success. At the national scale monitoring data already provide an overview of the ecological status of aquatic ecosystems, at least in some countries, while at the European level the options provided by the data still need to be fully exploited.

The value of monitoring *per se* is in analysing trends over time. Presently, the spatial resolution of WFD monitoring data is high, though somewhat different between European countries. As the first phase of monitoring has just ended, there is yet no assessment of trends; the monitoring data will be important both for judging short-term effects of individual restoration measures and for analysing long term trends. The particular value of the WFD monitoring data lies in the combination of a high spatial and a moderate temporal resolution.

Many European countries had a long tradition in biological monitoring of rivers; consequently, river assessment methods are now relatively well developed and intercalibrated However, rivers are very diverse and complex systems and assessment systems are often less predictable compared to those developed for lakes and coastal/transitional waters. At the same time rivers may provide deeper insight into causes of degradation, which are more complex due to the greater role of hydromorphological stress.

Much has been achieved with the implementation of the WFD, but many challenges remain to make optimal use of the unique monitoring data being acquired in order to achieve a maximum improvement in the ecological quality of European surface waters.

Acknowledgements

This paper is a result of the project WISER (Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery) funded by the European Union under the 7th Framework Programme, Theme 6 (Environment including Climate Change) (contract No. 226273), www.wiser.eu. We appreciate the detailed and helpful comments of an anonymous reviewer who greatly contributed to improving the paper.

References

Apitz SE, Elliott M, Fountain M, Galloway TS. European Environmental Management: Moving to an Ecosystem Approach. Integrated Environmental Assessment & Management 2006; 2(1): 80-85.

Arle J, Wagner F. Die Bedeutung der Gewässerstruktur für das Erreichen des guten ökologischen Zustands in den Fließgewässern des Freistaates Thüringen. Limnologie Aktuell: in press.

Aroviita J, Mykrä H, Hämäläinen. River bioassessment and the preservation of threatened species: towards acceptable biological quality criteria. In: Predictive models in assessment of macroinvertebrates in boreal rivers. Aroviita J, PhD Thesis, University of Jyväskylä, 2009; 201.

Aroviita J, Mykra H, Muotka T, Hamalainen H. Influence of geographical extent on typology- and model-based assessments of taxonomic completeness of river macroinvertebrates. Freshwater Biol 2009; 54: 1774-1787.

Blomqvist M, Cederwall H, Leonardsson K, Rosenberg R. Bedömningsgrunder för kust och hav. Bentiska evertebrater 2006. Rapport till Naturvårdsverket 2007-04-11. 70 pp. (in Swedish with English summary).

Borja A. The European Water Framework Directive: a challenge for nearshore, coastal and continental shelf research. Cont Shelf Res 2005; 25(14): 1768-1783.

Borja A. La investigación marina en las nuevas políticas europeas de gestión integrada. In: Gestión integrada de zonas costeras. Ed. JL Domenech. AENOR (Asociación Española de Normalización y Certificación), Madrid, in press: 407-455.

Borja A, Elliott M. What does ‘good ecological potential’ mean, within the European Water Framework Directive? Mar Pollut Bull 2007; 54: 1559-1564.

Borja A, Heinrich H. Implementing the European Water Framework Directive: The debate continues … Mar Pollut Bull 2005; 50: 486-488.

Borja A, Franco J, Valencia V, Bald J, Muxika I, Belzunce MJ, Solaun O. Implementation of the European Water Framework Directive from the Basque Country (northern Spain): a methodological approach. Mar Pollut Bull 2004; 48(3-4): 209-218.

Borja A, Muxika I, Franco J. Long-term recovery of soft-bottom benthos following urban and industrial sewage treatment in the Nervión estuary (southern Bay of Biscay). Mar Ecol-Prog Ser 2006; 313: 43-55.

Borja A, Josefson AB, Miles A, et al. An approach to the intercalibration of benthic ecological status assessment in the North Atlantic ecoregion, according to the European Water Framework Directive. Mar Pollut Bull 2007; 55 (1-6): 42-52.

Borja A, Bricker SB, Dauer, DM, Demetriades, NT, Ferreira JG, Forbes AT, Hutchings, P, Jia X, Kenchington R, Marques JC, Zhu C. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Mar Pollut Bull 2008a; 56: 1519-1537.

Borja A, Tueros I, Belzunce MJ, Galparsoro I, Garmendia JM, Revilla M, Solaun O, Valencia V. Investigative monitoring within the European Water Framework Directive: a coastal blast furnace slag disposal, as an example. J Environ Monitor 2008b; 10: 453-462.

Borja A, Miles A, Occhipinti-Ambrogi A, Berg T. Current status of macroinvertebrate methods used for assessing the quality of European marine waters: implementing the Water Framework Directive. Hydrobiologia 2009a; 633(1): 181-196.

Borja A, Bald J, Franco J, Larreta J, Muxika I, Revilla M, Rodríguez JG, Solaun O, Uriarte A, Valencia V. Using multiple ecosystem components, in assessing ecological status in Spanish (Basque Country) Atlantic marine waters. Mar Pollut Bull 2009b; 59: 54-64.

Borja A, Ranasinghe A, Weisberg SB. Assessing ecological integrity in marine waters, using multiple indices and ecosystem components: Challenges for the future. Mar Pollut Bull 2009c; 59: 1-4.

Borja A, Muxika I, Rodríguez JG. Paradigmatic responses of marine benthic communities to different anthropogenic pressures, using M-AMBI, within the European Water Framework Directive. Marine Ecology 2009d; 30: 214-227.

Brunke M, Lietz J. Regenerationsmaßnahmen und der ökologischer Zustand der Fließgewässer in Schleswig-Holstein. Limnologie Aktuell: in press.

Cardoso AC, Solimini A, Premazzi G, Carvalho L, Lyche A, Rekolainen S. Phosphorus reference concentrations in European lakes. Hydrobiologia 2007; 584: 3-12

Carstensen J. Statistical principles for ecological status classification of Water Framework Directive monitoring data. Mar Pollut Bull 2007; 55: 3–15.

Carvalho L, Phillips G, Maberly S and Clarke R. Chlorophyll and Phosphorus Classifications for UK Lakes. Final Report to SNIFFER (Project WFD38), Edinburgh, October 2006, 81 pp.

Carvalho L, Dudley B, Dodkins I, Clarke R, Jones I, Thackeray S and Maberly S. Phytoplankton Classification Tool (Phase 2). Final Report to SNIFFER (Project WFD80), Edinburgh, June 2007, 94 pp

Carvalho L, Solimini A, Phillips G, van den Berg M, Lyche-Solheim A, Mischke U, Pietiläinen O-P, Poikane S, Tartari G. Site-specific chlorophyll reference conditions for lakes in Northern and Western Europe. Hydrobiologia 2009; 633: 59-66.

CEN. Water quality – Guidance standard for the routine analysis of phytoplankton abundance and composition using inverted microscopy (Utermöhl technique) CEN TC 230/WG 2/TG 3/N83, May 2004.

CIS Working Group 2.2 on Heavily Modified Water Bodies. Toolbox on identification and designation of artificial and heavily modified water bodies (2003).

Clarke RT, Hering D. Errors and uncertainty in bioassessment methods – major results and conclusions from the STAR project and their application using STARBUGS. Hydrobiologia 2006; 566: 433-439.

Clarke RT, Wright JF, Furse MT. RIVPACS models for predicting the expected macroinvertebrate fauna and assessing the ecological quality of rivers. Ecol Model 2003; 160: 219-233.

Commission of the European Communities. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2008:164:0019:0040:EN:PDF

Commission of the European Communities. Commission staff working document. Accompanying document to the communication from the commission to the European Parliament and the Council 'Towards Sustainable Water Management in the European Union'; first stage in the implementation of the Water Framework Directive 2000/60/EC [COM(2007) 128 (2007).

Commission of the European Communities. Commission staff working document accompanying the report of the Commission to the European Parliament and the Council in accordance with article 18.3 of the Water Framework Directive 2000/60/EC on programmes for monitoring of water status. COM(2009)156 (2009).

Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Identification and Designation of Heavily Modified and Artificial Water Bodies. WFD CIS Guidance Document No. 4 (2002).

Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Guidance on the intercalibration process 2004 – 2006. WFD CIS Guidance Document No. 14 (2005).

Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Guidance document on Eutrophication assessment in the context of European water policies. WFD CIS Guidance Document No. 23 (2009).

De Jonge VN, Elliott M, Brauer VS. Marine monitoring: its shortcomings and mismatch with the EU Water Framework Directive's objectives. Mar Pollut Bull 2006; 53: 5-19.

Downes BJ, Barmuta LA, Fairweather PG, Faith DP, Keought MJ, Lake P., Mapstone BD, Quinn GP. Monitoring ecological impacts – concepts and practice in flowing waters. Cambridge University Press, Cambridge, UK (2002).

Duarte CM, Conley DJ, Carstensen J, M Sánchez-Camacho. Return to Neverland: Shifting Baselines Affect Eutrophication Restoration Targets. Estuaries and Coasts 2009; 32: 29-36.

Dufour S, Piegay H. From the myth of a lost paradise to targeted river restoration: forget natural references and focus on human benefits. Riv Res Appl 2009; 25 (5): 568-581.

Elliott M, Quintino V. The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Mar Pollut Bull 2007a; 54: 640-645.

Ferreira J, Vale C, Soares C, Salas F, Stacey P, Bricker S, Silva M, Marques J. Monitoring of coastal and transitional waters under the E.U. Water Framework Directive. Environ Monit Assess 2007; 135: 195-216.

Friedrich G, Coring E, Küchenhoff B. Vergleich verschiedener europäischer Untersuchungs- und Bewertungsmethoden für Fließgewässer. Landesumweltamt Nordrhein-Westfalen, Essen, Germany, 1995.

Furse M, Hering D, Moog O, Verdonschot PFM, Sandin L, Brabec K, Gritzalis K, Buffagni A, Pinto P, Friberg N, Murray-Bligh J, Kokes J, Alber R, Usseglio-Polatera P, Haase P, Sweeting R, Bis B, Szoszkiewicz K, Soszka H, Springe G, Sporka F & Krno I. The STAR project: context, objectives and approaches. Hydrobiologia 2006; 566: 3-29.

Ghetti PF, Bonazzi G. A comparison between various criteria for the interpretation of biological data in the analysis of the quality of running waters. Water Res 1977; 11: 819–831.

Heiskanen AS, van de Bund W, Cardoso AC, Noges P. Towards good ecological status of surface waters in Europe - interpretation and harmonisation of the concept. Water Sci Technol 2004; 49: 169-177.

Hering D., Moog O, Sandin L, Verdonschot PFM. Overview and application of the AQEM assessment system. Hydrobiologia 2004; 516: 1-20.

Hering D., Johnson RK, Buffagni A. Linking organism groups - major results and conclusions from the STAR project. Hydrobiologia 2006; 566: 109-113.

Hoornbeek JA. Policy-making institutions and water policy outputs in the European Union and the United States: A comparative analysis. Journal of European Public Policy 2004; 11(3): 461-496+567.

Hull SC, Freeman SM, Rogers SI, Ash J, Brooke J, Elliott, M. Methodology for the Provisional Identification and Formal Designation of Heavily Modified Water Bodies in UK Transitional and Coastal Waters under the EC Water Framework Directive. Environment Agency R&D Technical Report, Bristol UK, 2004.

Jähnig SC, Lorenz AW, Hering D. Restoration effects, Habitat mosaics and macroinvertebrates – does channel form determine community composition? Aquat Conserv, 2009; 19: 157–169.

Jeppesen E, Søndergaard M, Jensen JP, Havens K, Anneville O, Carvalho L, Coveney MF, Deneke R, Dokulil M, Foy B, Gerdeaux D, Hampton SE, Kangur K, Köhler J, Körner S, Lammens E, Lauridsen TL, Manca M, Miracle R, Moss B, Nõges P, Persson G, Phillips G, Portielje R, Romo S, Schelske CL, Straile D, Tatrai I, Willén E, Winder M. Lake responses to reduced nutrient loading – an analysis of contemporary long-term data from 35 case studies. Freshwater Biol 2005; 50: 1747–1771.

Jones HP, Schmitz OJ. Rapid recovery of damaged ecosystems. PLoS ONE; 2009, 4(5): 1–6.

Kail J, Hering D. Using large wood to restore streams in Central Europe: Potential use and likely effects. Landscape Ecol 2005; 20: 755-772.

Kail J, Wolters C. Analysis and evaluation of large-scale river restoration planning in Germany to better link river research and management. River Research and Applications, in press.

Kampa E, Kranz N. WFD and Hydromorphology. European Workshop, 17-19 October 2005, Prague, workshop summary report. 2005, available from http://www.ecologic-events.de/hydromorphology/documents/967_summary.pdf

Kauppila P, Weckström W, Vaalgamaa S. Tracing pollution and recovery using sediments in an urban estuary, northern Baltic Sea: are we far from ecological reference conditions? Marine Ecology Progress Series 2005; 290: 35-53

Krause-Jensen D, Carstensen J, Dahl K, Bäck S, Neuvonen S. Testing relationships between macroalgal cover and Secchi depth in the Baltic Sea. Ecol Ind 2009; 9: 1284-1287.

Lammens E, van Luijn F, Wessels Y, Bouwhuis H, Noordhuis R, Portielje R, van der Molen D. Towards ecological goals for the heavily modified lakes in the IJsselmeer area, The Netherlands. Hydrobiologia 2008; 599: 239–247.

Lepistö L, Vuorio K, Holopainen A-L, Palomäki A, Järvinen M, Huttunen M. Quality control in phytoplankton analysis. The Finnish Environment 2009; 40: 1-31 (in Finnish, summary in English) (ISSN 1796-1637).

Lorenz A, Feld CK, Hering D. Typology of streams in Germany based on benthic invertebrates: Ecoregions, zonation, geology and substrate. Limnologica 2004; 34(4): 390-397.

Lyche-Solheim A., Rekolainen S, Moe SJ et al. Ecological threshold responses in European lakes and their applicability for the Water Framework Directive (WFD) implementation: synthesis of lakes results from the REBECCA project. Aquat Ecol 2008; 42: 317-334.

Lyche-Solheim A, Bouraoui F, Grizzetti B, Collins R, Prchalova H, Moe J, Globevnik L, Kodes V, Selvik JR, Morabito G, Løvik JE, Hobæk A. Freshwater eutrophication assessment for the State-of-Environment Report 2010. EEA-ETC (http://water.eionet.europa.eu/ETC_Reports).

McLusky DS, Elliott M. The Estuarine Ecosystem; ecology, threats and management, 3rd edition, OUP, Oxford, 2004, 216 pp.

McLusky DS, Elliott M. Transitional Waters: a new approach, semantics or just muddying the waters? Estuar Coast Shelf S 2007; 71: 359-363.

Mee LD, Jefferson RL, Laffoley Dd'A, Elliott M. How good is good? Human values and Europe's proposed Marine Strategy Directive. Mar Pollut Bull, 2008; 56: 187-204.

Moe SJ, Dudley B, Ptacnik R. REBECCA databases: experiences from compilation and analyses of monitoring data from 5,000 lakes in 20 European countries. Aquatic Ecology, 2008; 42: 183-201.

Moss B. Shallow lakes, the water framework directive and life. What should it all be about? Hydrobiologia 2007; 584: 381-394.

Moss B. The Water Framework Directive: Total environment or political compromise? Sci Total Environ 2008; 400 (1-3): 32-41

Moss B, Stephen D, Alvarez C, et al. The determination of ecological status in shallow lakes - a tested system (ECOFRAME) for implementation of the European Water Framework Directive. Aquat Cons 2003; 13 (6): 507-549.

Muxika I., Borja A., Bald J. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. Mar Pollut Bull 2007; 55: 16-29

OIWater. Bilan sur la surveillance des cours d'eau - Tome 1 : les efforts de surveillance et de bancarisation des données relatives é la qualité (River monitoring in France). Onema, France, 2009 ; 61 pp.

Palmer M, Menninger H, Bernhardt E. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater Biol, in press.

Peeters ETHM, Franken RJM, Jeppesen E, Moss B, Becares E, Hansson LA, Romo, Kairesalo T, Gross EM, van Donk E, Noges T, Irvine K, Kornijow R, Scheffer M. Assessing ecological quality of shallow lakes: Does knowledge of transparency suffice? Basic and Applied Ecology 2009; 10: 89-96.

Perus J, Bonsdorff E, Bäck S, Lax H-G, Westberg V, Villnäs A. Zoobenthos as indicator of ecological status in coastal brackish waters: A comparative study from the Baltic Sea. Ambio 2007; 36 (2-3): 250-256.

Pont D, Bady P, Logez M, Veslot J. EFI+ Project. Improvement and spatial extension of the European Fish Index Deliverable 4.1 : Report on the modelling of reference conditions and on the sensitivity of candidate metrics to anthropogenic pressures. Deliverable 4.2: Report on the final development and validation of the new European Fish Index and method, including a complete technical description of the new method. 6th Framework Programme Priority FP6-2005-SSP-5-A. N° 0044096. Final Report, 179pp. (http://efi-plus.boku.ac.at/), 2009.

Pont D, Hugueny B, Beier U, Goffaux D, Melcher A, Noble R, Rogers C, Roset N, Schmutz S. Assessing river biotic condition at the continental scale: a European approach using functional metrics and fish assemblages. J Appl Ecol 2006; 43: 70-80.

Pont D, Hugueny B, Rogers C. Development of a fish-based index for the assessment of "river health" in Europe: the European Fish Index (EFI). Fisheries Manag Ecol 2007; 14: 427-439.

Sandin L. Testing the EC Water Framework Directive "one-out, all-out" rule - simulating different levels of assessment errors along a pollution gradient in Swedish streams. Verh Internat Verein Limnol 2005; 29: 334-336.

Sandin L, Verdonschot PFM. Stream and river typologies - major results and conclusions from the STAR project. Hydrobiologia 2006; 566: 33-37.

Schmutz S, Cowx IG, Haidvogl G, Pont D. Fish-based methods for assessing European running waters: a synthesis. Fisheries Manag Ecol 2007; 14: 369-380

Schmidt-Kloiber A, Graf W, Lorenz A, Moog O. The AQEM/STAR taxalist - a pan-European macro-invertebrate ecological database and taxa inventory. Hydrobiologia 2006; 566: 325-342.

Sondergaard M, Jeppesen E, Jensen JP, Amsinck SL. Water framework directive: Ecological classification of danish lakes. J Appl Ecol 2005; 42: 616-629.

Stubauer I, Moog O. Taxonomic sufficiency versus need for information – comments based on Austrian experience in biological water quality monitoring. Verh Internat Verein Limnol 2000; 27: 1–5.

Tueros I, Borja A, Larreta J, Rodríguez JG, Valencia V, Millán E. Integrating long-term water and sediment pollution data, in assessing chemical status within the European Water Framework Directive. Mar Pollut Bull 2009, 58(9): 1389-1400.

Uriarte A, Borja A. Assessing fish quality status in transitional waters, within the European Water Framework Directive: Setting boundary classes and responding to anthropogenic pressures. Estuar Coast Shelf S 2009; 82: 214-224.

Verdonschot PFM. Typology of macrofaunal assemblages – a tool for the management of running waters in the Netherlands. Hydrobiologia 1995; 297: 99-122.

WG ECOSTAT. Overall approach to the classification of ecological status and ecological potential. 2003, version 4, 45pp.

WG ECOSTAT. Guidance on the Intercalibration Process 2008-2011 [version 7.0]. 2009, 53 pp.

Table 1: Overview of successes and problems encountered in the implementation process of the Water Framework Directive related to ecological assessment of water bodies, of causes, consequences, already applied solutions and recommendations. Abbreviations: HMWB: Heavily Modified Water Bodies; BQE: Biological Quality Elements; WFD: Water Framework Directive; RBMP: River Basin Management Plans; EEA: European Environment Agency; CIS: Common Implementation Strategy; WISE: Water Information System for Europe; SEIS: Shared Environmental Information System; MSFD: Marine Strategy Framework Directive.

Issue	Successes	Problems encountered	Already applied or initiated solutions	Future recommendations
Assessment of ecological status				
National assessment systems	− Assessment systems reflecting different stressors for most BQEs and water types now available, adapted to the needs of member states − Transparent development process involving scientists, water managers and stakeholders	− Effort and long time period required for development − Degree of complexity of some assessment systems − Different and partly incomparable systems by member states − Lack of data for developing indicators of some widespread pressures (e.g. hydromorphology) − Lack of reference sites in Central and Mediterranean Europe	− Intercalibration of national assessment systems	− Further improvement and harmonisation of assessment systems based on experiences of first cycle of intercalibration and monitoring
Uncertainty in assessment	− Principle of giving status classifications as probabilities best developed to reflect sources of sampling and analysis variability − Simple underlying statistical principles developed − Stimulated pan-European training in identification	− Only few assessment systems have included uncertainty estimation − Communication of the concept of uncertainty to water managers − Due to data constraints, less developed for assessing uncertainty due to temporal variability	− For selected assessment systems: quantification of sources of variability, e.g. sampling and identification error	− Standardised approach for uncertainty estimation for all assessment systems − Improved training in sampling and identification and further standardisation of biological recording to minimise sources of error − Restrict sampling to one season if possible, to reduce

Issue	Successes	Problems encountered	Already applied or initiated solutions	Future recommendations
				natural variability
Typology	– Typologies or prediction systems have been developed by all member states – Developed typologies enable higher precision of ecological assessment	– Need to find the balance between being too specific (too many types) and being too general (types do not sufficiently reflect natural variability)	– Broadly defined types for rough ecological assessment (e.g. 'common types' used for intercalibration – Improved typology for some of the 'Geographical Intercalibration Groups' – Improved prediction models to overcome general problems of typologies	– Improve site-specific assessment models (prediction systems), once sufficient data are available, esp. for sites close to type boundaries –
Intercalibration	– Methods for intercalibration were developed – Intercalibration was successfully completed for several BQEs and water types – Many ssessment schemes now intercalibrated have comparable class boundaries	– Differences in national assessment systems, due to biomonitoring traditions – Original WFD approach for intercalibration (small number of sites representing class boundaries) was not feasible – Effort and time required for intercalibration has been more than expected – Dissemination of intercalibration approaches and results	– Intercalibration methods based on 'common metrics' – New intercalibration guidance to ensure more consistent ways to compare, evaluate and adjust the assessment systems (intercalibration approaches)	– Increased effort to disseminate the need for intercalibration – Clearer guidelines on robustness/uncertainty of metrics to be included in intercalibration
Combination of assessment results ('one-out all-out principle')	– Reduced type II errors (water body is falsely classified as good or high), in line with the precautionary principle – Sufficient protection of most sensitive BQE for different pressures	– Increased type I error (water body is falsely classified as moderate or worse), risk of applying measures where they are not really needed		– Estimate the degree of type I and type II errors for each assessment system – Improve metrics and monitoring programmes to minimise variability. – Skip metrics and BQEs with too high variability – Consider other combination rules in future amendments of the WFD
Assessment of Heavily Modified Water Bodies (HMWB)	– Application of appropriate quality targets which can be	– HMWBs have not been regarded in many assessment		– Assessment of HWMB should be based on the same

Issue	Successes	Problems encountered	Already applied or initiated solutions	Future recommendations
	achieved following restoration – Two well-suited approaches for assessing HMWB available (CIS approach and Prague approach)	systems – No agreement yet on which approach should be primarily used		metrics as for natural water bodies
Monitoring systems				
Monitoring data	– Huge amounts of data on aquatic communities is being collected (useful for many purposes) – Sampling and assessment systems are standardised within countries and sometimes between countries – Following intercalibration ecological status classes are comparable between member states	– Comparability of original data between countries is limited due to different sampling methods, taxonomic resolution and density of sampling sites – Original data are not centrally stored – Monitoring focused on biological structure, not on function or ecosystem services	– Establishment of a Europe-wide central monitoring network composed of selected surveillance monitoring sites (e.g. linked to EEA EIONET or WISE)	– Links of national databases to central systems such as WISE to increase accessibility of data
Surveillance monitoring and operational monitoring	– Surveillance monitoring and operational monitoring are being used effectively to fulfil WFD purposes – Programmes for long-term monitoring (surveillance monitoring) and for planning restoration (operational monitoring) are available in most countries	– Very few surveillance monitoring sites in many member states, which limits European State-of-Environment overviews, as well as the detection of emerging stressors and long-term trends – No Europe-wide data base on surveillance monitoring		– Establishment of a Europe-wide central monitoring network composed of selected surveillance monitoring sites (e.g. linked to EEA EIONET or WISE)
Monitoring requirements of WFD and other European legislation	– WFD filled important gaps in surface water monitoring and management	– Definitions of objectives and requirements of WFD and other directives are not always consistent – Potential synergies of monitoring systems resulting from different directives not fully exploited	– Guidance on Eutrophication (2009) recommending how to read across different directives and conventions recently published presenting a harmonisation of the different objectives	– Clear geographical definition where the WFD ends and where the MSFD starts – Exploring and using synergies of monitoring for different directives for other pressures than eutrophication
River Basin Management Plans (RBMPs)				

Issue	Successes	Problems encountered	Already applied or initiated solutions	Future recommendations
Bridging ecology and management in RBMPs	− Management decisions are based on ecological effects of stressors on structure rather than on the stressor itself − Plans are drafted for entire catchments, irrespective of administrative borders	− Deriving management decisions from ecological data are difficult in case of complex multi-stressor situations − Results of ecological assessment were often not available in time for the first version of RMPBs − How stressors and biological structure affect ecosystem services is not well understood − Some metrics are not related to specific pressures (general degradation metrics) and are difficult to apply to plan restoration measures		− Make dose-response relationships between stressors and the biotic response available well before the design of the second cycle of River Basin Management Plans (concerning the effects of degradation and of restoration) − Consider further development of functional indicators that reflect ecosystem services − Develop political instruments that will guarantee enforcement of RBMPss
'Good status' as general quality target	− Generally applicable target for all 'natural water bodies' in all member states	− High status sites may play a key role for maintaining aquatic biodiversity	− WFD prohibits the deterioration of ecological status, including the degradation of high status sites to good status sites	− Establishing a network of 'high status sites' as key areas for protecting aquatic biodiversity, and to ensure ecosystem services for all types of water bodies
Ecological status response to restoration	− Stimulated synthesis of experiences on biotic responses to traditional restoration measures (oligotrophication, pollution control)	− Response of biota to restoration measures in complex multi-stressor situations poorly predictable − Lack of data and experience on spatial and temporal scales required for restoration	− Judging restoration success through operational monitoring	− Dedicated monitoring of a subset of restoration sites with a higher spatial and temporal resolution both before and after restoration measures are implemented − Long-term monitoring of restoration measures to analyse spatial and temporal requirements of ecosystems to recover
Ecological and political timescales	− Clear goal to reach good ecological status for all water	− Implementation and success of restoration measures	− Consider direction towards goals when assessing	− Disseminate results and expectations concerning the

Issue	Successes	Problems encountered	Already applied or initiated solutions	Future recommendations
	bodies by 2015 (extension to 2027 possible) – RMBPs are developed accordingly	requires long time periods – Insufficient knowledge on how fast biota will respond to restoration – Long time needed to implement measures that require land use change – Time lags due to internal nutrient loading and low recolonisation potential expected	restoration success, not simply whether target is attained or not	time spans required for recovery to avoid frustration of water managers – Prioritisation of measures concerning the recolonisation potential
Emerging stressors	– WFD principle of bioassessment (comparing observed and expected community) reflects potentially the impact of all stressors	– Assessment metrics often focussed on 'traditional stressors' (organic pollution, eutrophication) – No metrics for the effects of emerging stressors (climate change, siltation, alien species) included	– Research examining impacts of climate change on reference conditions – WFD-CIS Guidance on how to handle climate change and alien species are drafted and will soon become available	– Exploring response trajectories and resilience of metrics – Keeping assessment systems flexible and adding metrics specific for emerging stressors (such as temperature preferences for climate change effects)

Table 2: Rivers in mountainous regions and lowlands of Germany: Percentage of sites classified as moderate, poor or bad by single organism groups and by combinations of organism groups.

	Mountains	Lowlands
Diatoms (n = 865)	64%	68%
Invertebrates (n = 1,552)	66%	80%
Fish (n = 187)	63%	78%
Invertebrates and fish (n = 178)	86%	92%
Diatoms and invertebrates (n = 765)	80%	91%

Table 3. Relationships among the different European environmental directives, conventions and legislation addressing surface water bodies, regarding their application level and objectives, from the lowest (bottom) to the highest spatial and complexity level (up) (modified from Borja, in press).

Application level	Objectives/ecological basis	Legislation
Global	The Ecosystem Approach, sustainability	UNCED, UNCLOS, IMO, CBD
Europe/ ecoregions	Ecosystem-based management, ecological integrity	Water Framework Directive, Marine Strategy Framework Directive
Uses/Sectoral policy	Thematic strategies	Urban wastewater treatment directive, Nitrates Directive, Common Agricultural Policy, Renewable Energy Directive, Drinking Water Directive, Bathing Water Directive, Fisheries Common Policy, Maritime Policy
Regional seas	Quality and uses, from sectoral (pollution) to ecosystem-based approach	International Conventions (MARPOL, HELCOM, OSPAR, Barcelona)
River basins	Chemical and ecological quality status	Water Framework Directive
Ecosystems	Ecological processes, ecological status	Water Framework Directive, Marine Strategy Framework Directive, Recommendation on Integrated Coastal Zone Management
Habitats	Habitat networks, connectivity, habitat protection	Habitats Directive, Water Framework Directive, Recommendation on Integrated Coastal Zone Management

Species	Habitat quality, biodiversity protection	Habitats Directive, Birds Directive