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Viewpoint

The use of benthic indicators in Europe: From the Water Framework Directive to the Marine Strategy Framework Directive

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ABSTRACT

The Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD) are the European umbrella regulations for water systems. It is a challenge for the scientific community to translate the principles of these directives into realistic and accurate approaches. The aim of this paper, conducted by the Benthos Ecology Working Group of ICES, is to describe how the principles have been translated, which were the challenges and best way forward. We have tackled the following principles: the ecosystem-based approach, the development of benthic indicators, the definition of 'pristine' or sustainable conditions, the detection of pressures and the development of monitoring programs. We concluded that testing and integrating the different approaches was facilitated during the WFD process, which led to further insights and improvements, which the MSFD can rely upon. Expert involvement in the entire implementation process proved to be of vital importance.

1. Introduction

In Europe, the umbrella regulations for addressing the ecological quality of the water systems are the Water Framework Directive (WFD; 2000/60/EC), for lakes, rivers, transitional (=estuaries and lagoons) and coastal waters (Table 1), and the Marine Strategy Framework Directive (MSFD; 2008/56/EC) for marine waters (Table 2). The ecological concept behind both directives is, in principle, very simple, and consists of comparing the current state of an area with that which would be expected under minimal or sustainable human use of that area and, in case of degradation, intervening to bring it back to the desired good status (Mee et al., 2008). For the WFD, a variety of indicators, target values and reference setting approaches for assessing good ecological status (GES) has been developed, intercalibrated, discussed and published during the last decade, and the process continues (Borja et al., 2009d; Hering et al., 2010). Implementation of the recent MSFD has started, by defining the criteria/indicators of the eleven qualitative descriptors for assessing good environmental status (GEnS) (Table 2). From the WFD process, we learned that defining GES and the translation of it into a set of measurable environmental targets and associated indicators is not an easy task (Hering et al., 2010). This process is done for the WFD at the member state level, and therefore required intercalibration of the entire process between the member states of certain geographic regions. Within the MSFD, this process is regionalized, because the countries per regional sea have to define common indicators per descriptor for GEnS (Salomon, 2006; Rice et al., 2010).

Both directives are similar in concept and lessons learned from the WFD implementation process will help in implementing the MSFD. It is widely acknowledged among the scientific community that none of the existing approaches is yet perfect and that the realization of the principles within the directives is based on currently available scientific knowledge. Therefore, the Benthos Ecology Working Group (BEWG) part of the International Council

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Table 1
Water Framework Directive (WFD).

The European Water Framework Directive (WFD; 2000/60/EC) aims at achieving 'good' ecological and chemical quality status for all water types, by 2015. The quality status of a water body can be determined based on the evaluation of biological quality elements, which are phytoplankton, macroalgae, macro-invertebrates and fish (the latter only in transitional waters), which are supported by chemical, physico-chemical (e.g. transparency, thermal and oxygen conditions, salinity and nutrients) and hydromorphological (e.g. depth variation, quantity structure and substrate of the sub-tidal and intertidal zone, tidal regime) quality elements. GES is defined as 'the values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions'. The evaluation of GES is based on the integration of well defined quality criteria per quality element. Each of these quality criteria supports a classification (bad, poor, moderate, good and high) to measure the 'health' of the system compared to reference conditions. For the biological quality element macro-invertebrates, the composition and abundance of the fauna has to be identified. Within a WFD context, many benthic indicators were developed and intercalibrated, which combine some benthic variables such as abundance, biomass, diversity (e.g. Shannon Wiener, Margalef, Simpson indexes), Bray-Curtis similarity, species sensitivity/tolerance classifications (e.g. AMBI, $ES50_{0.05}$ species values) in a multivariate or multimetric way (Borja et al., 2007)

Table 2
Marine Strategy Framework Directive (MSFD).

The main objective of the Marine Strategy Framework Directive (MSFD; 2008/56/EC) is to achieve good environmental status (GEnS), by 2020. GEnS is defined as 'the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations'. Therefore, the MSFD established a framework for the development of strategies designed to achieve GEnS, which takes into account the structure, function and processes of the marine ecosystems together with natural physiographic, geographic and climatic factors, as well as physical and chemical conditions including those resulting from human activities in the area concerned. This is reflected in the development of 11 quality descriptors for determining good environmental status, for which a set of criteria and associated indicators were proposed by expert groups. Based on the expert group reports, an EU Commission Decision document on the criteria and methodological standards on GEnS of marine waters were defined (2010/477/EU). Based on this document, macro-invertebrates were taken into account in four descriptors:
• 'Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.' Proposed criteria are species distribution, population size, population condition, habitat distribution, extent and condition for the benthic community among other fauna groups (Table 1 of Annex III of MSFD).
• 'Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystem.' In case benthic species were identified as non-indigenous, in particular invasive, this descriptor has to be taken into account by GEnS evaluation.
• 'All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.' A proposed indicator for the criterion 'abundance/distribution of key groups/species' are abundance trends of functionally important selected groups/species (e.g. biological groups with high turnover rate, habitat defining groups/species, ...).
• 'Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected.' For the criterion on the condition of benthic community, the following indicators are proposed: presence of particularly sensitive/tolerant species, multi-metric indexes assessing benthic community condition and functionality (species diversity and richness, proportion of opportunistic to sensitive species), proportion of biomass of number or individuals above some specified length/size class and parameters (slope and intercept) of the size spectrum of the aggregate size composition data.
For a number of criteria and related indicators, the need for further development and additional information was identified in the document. Member States need to consider each of the criteria and related indicators in order to identify those which are to be used to determine the GEnS. Methodological standards still need to be developed.

for the Exploration of the Sea (ICES) decided to compile this viewpoint paper with the objective of highlighting key issues related to the fulfillment of the principles of both directives, with a focus on benthic macro-invertebrates.

Several challenges were encountered during the WFD implementation, e.g. the development of assessment methods and the implementation of assessment systems in monitoring programs (Borja et al., 2009d). Therefore, this paper focuses on how the principles underlying both directives have been translated into implementable approaches, on some of the approaches adopted as part of the WFD and what these approaches mean for the implementation of the MSFD. First we considered the use of the 'ecosystem approach' principle in both directives. Second, we addressed, as they relate to the definition of GES and GEnS, on the development of benthic indicators for classification, definition of 'pristine' or sustainable conditions and the importance of relating ecological measurements to pressures. In this case, we discussed the problems related to detecting different anthropogenic impact types, distinguishing between anthropogenic versus natural changes using indicators and how to evaluate the pressure "non-indigenous or alien species". Third, we addressed on monitoring programs (effort and quality), which have to provide sufficient information to allow a confident assessment of GES and GEnS. For each principle, the BEWG formulated advice on how to proceed in the future (Table 3).

2. Use of the ecosystem approach

To apply an ecosystem-based approach is considered one of the most important requirements for sustainable environmental management and was defined as 'a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way' (United Nations Convention on Biological Biodiversity, 29 December 1993). An ecosystem-based approach is emerging for the assessment and management of systems utilizing strategies for linking science-based assessments of the changing states of ecosystems to socio-economic benefits (goods and services) expected from achieving long-term sustainability of their resources (Sherman and Duda, 1999; Rosenberg and Mcleod, 2005; Leslie and Mcleod, 2007). To manage human pressures on marine environments, recent and worldwide approved, legislative instruments address the need to assess a system's condition (Borja and Dauer, 2008). The concept of determining a system's health has to take into account the structure, function and processes of marine ecosystems bringing together natural physical, chemical, physiographic, geographic and climatic factors, and then integrate these with any human activities and impacts in the area concerned (Borja et al., 2009b). This approach is partly used in the WFD, where a few biological elements and supporting physico-chemical parameters, along with the concentration of pollutants are selected to assess

Table 3
Summary table of how some key principles of the Water Framework Directive and Marine Strategy Framework Directive were filled in, the associated challenges and the way forward suggested by the BEWG. Note: GEnS: Good Environmental Status.

Principles		Water Framework Directive	Marine Strategy Framework Directive
2. Use of the Ecosystem approach	Realization	• Biological quality elements with supporting chemical, physico-chemical and hydromorphological variables	• Eleven descriptors, with several indicators covering ecological, physical, chemical and anthropogenic components of the ecosystem
	Challenge	• Integration of the elements based on one out - all out principle', which is not always appropriate	• Selection of the appropriate indicators and the integration of the several indicators per descriptor
	Way to go	• Scientific selection of elements/indicators in relation to their sensitivity, robustness and confidence. • Integration of indicators based on a decision tree process, with a clear transparency of the integration acts at indicator and descriptor level.	
3.1 Benthic Indicators	Realization	• National approaches, which require intercalibration • Multi-metric benthic indicators	• Regional approach, with common indicators • Mainly univariate indicators per descriptor
	Challenge	• Comparability of the national approaches	• No comparability tests on indicator level needed, but still on other aspects of the Directive (e.g. GEnS thresholds) • Sensitivity of single univariate benthic indicators less clear!
	Way to go	• The selection of appropriate indicators, with complementary properties and related to the Directive objectives. • Integration of single univariate indicators required to detect the complex response of benthos	
3.2 From pristine conditions to sustainably functioning ecosystems	Realization	• Reference = 'undisturbed (pristine) condition', to be determined based on reference sites or benchmarking	• Reference = 'sustainable functioning ecosystems', but no methodology for determining thresholds for GEnS
	Challenge	• No benthic reference sites, poor historical data • Expert judgment good first step, but needs funding	• What is good or sustainable? • No single GEnS thresholds for any indicator will be appropriate within a region
	Way to go	• The use of clear stressor-response data • Defining the 'naturalness' of the system	
3.3.1 Anthropogenic pressure types	Realization	• Indicators have to prove their pressure type dependency	• Indicators have to be selected based on pressure type (most appropriate, measurable)
	Challenge	• Multi-pressure environments • Large scale pressures • No impact free areas	
	Way to go	• Marine Protected Areas (MPAs) • Accurate and detailed quantification of the pressure types in the marine systems	
3.3.2 Natural versus anthropogenic response	Realization	• Indicators not or less sensitive to natural variability	
	Challenge	• Currently less investigations regarding sensitivity of indicators to natural variability and scoping the natural variability in defining reference conditions	• Availability of detailed data on large temporal and spatial scale
	Way to go	• Integration of all available temporal and spatial data information • The use of spatially well designed monitoring systems	
3.3.3 Alien species or non-indigenous species	Realization	• Alien species were considered as a pressure	• Non-indigenous species is a descriptor
	Challenge	• May not be present at high status. • Measures to remove or reduce the impact are scarce	• Measures to remove or reduce the impact are scarce
	Way to go	• Research has to focus on the effect of alien species (function, niche) on the ecosystem • Prevention of further invasions by early warning systems (precautionary principle)	
4. Monitoring requirements for environmental assessment	Realization	• Monitoring programs on national level	• National monitoring programs to be integrated on regional sea or sub-sea level
	Challenge	• Influence of sampling strategy type on assessment results • Diversity of national approaches in sampling strategy	
	Way to go	• Use of the habitat approach (stratified sampling strategy) in benthos monitoring • Incorporation of statistical power, effect size and variance determination in determining number of samples • Setting of an adequate time scaling of the monitoring in relation to the indicator type • Use of standard benthic quality assurance guidelines • Adaptation of national monitoring programs towards cost-effective, integrative strategies • A switch from 'station oriented monitoring' towards a 'basin or system oriented monitoring'	

the health of the ecosystem (Table 3) (Borja and Dauer, 2008; Borja et al., 2009c). To these structural components may be added other ecosystem attributes, such as food web dynamics, species diversity, and the distribution of life histories. These are not direct biological properties but rather functions of the entire ecosystem (Weisberg et al., 1997; Fulton et al., 2005; Rogers et al., 2007; Lavesque et al., 2009; Samhouri et al., 2009). These ecosystem attributes are important since they provide information about the functioning and status of the ecosystem, and have been widely perceived as potentially useful indicators of environmental status (Borja et al., 2009d; Samhouri et al., 2009). They are intended to facilitate assessments of GES or GEnS at the ecosystem level ('ecosystem-based approach' or 'holistic approach' methodologies). This means a step forward from the (structural) community level assessment to the (functional) ecosystem level assessment, similar to the leap from the individual species level to the community level, as the ecosystem is more than the mere sum of physical, chemical and biological elements. The ecosystem-based approach is taken into account in more detail for the MSFD than for the WFD, by defining a variety of physical, chemical and biological criteria (all eco-system components, functional traits) for each descriptor (Borja et al., 2009d; Rice et al., 2010). This represents a challenge in the evaluation of ecological integrity at the ecosystem level, using all information available and including as many elements, indicators and parameters as reasonable. The selection of the criteria per descriptor that has to be conducted in each region or sub-region is still a major challenge for the implementation of the MSFD by each member state. An appropriate selection of elements and indicators for the specific goals of the directives, based on their sensitivity, robustness and confidence is necessary (Table 3). Accordingly, a balance has to be sought between the need for the ideal assessment covering all parts of the ecosystem (i.e. ideal world goals) and the feasibility of achieving this objective, due to e.g. practical and financial limitations (i.e. real world constraints), which is a major challenge for cost-effective implementation of the directives.

Furthermore, the selection of an appropriate set of indicators is one thing, the integration of all the indicators into a single score indicating status and performance of an aquatic system is another (Aubry and Elliott, 2006; Borja et al. 2008, 2009d; Foden et al., 2008). Simple approaches such as the 'one out, all out' principle (Borja, 2005) of the WFD, which scores the quality of a water body from the worst rated element, may be a useful starting point, but eventually should be avoided (Borja et al., 2009d). Using an averaging approach with weighting of the different indicators is also not ideal, due to the subjectivity and averaging out of indicators in low or high status. In addition, this is also not a useful approach when different indicators are used to express the ecological impact of different anthropogenic pressures or when methods used in the assessment are not reliable (Borja and Rodriguez, 2010). We suggest a decision tree, where the elements are weighted based on their confidence in assessing the status (e.g. benthos, with contrasted and intercalibrated methods) as a more accurate approach for global classification or their sensitivity to the pressures in the system (Borja et al., 2008). In the case of the MSFD, this will be a major challenge, due to the high number and variety of descriptors and indicators to assess GEnS of one region or sub-region in relation to the various human pressures there. Despite the fact that managers would like to have a single final score for GES and GEnS, it is advisable to report to the governments and public with good visibility of the assessments at indicator and descriptor level due to their difference in confidence and sensitivity.

3. Defining good ecological (WFD) or good environmental status (MSFD)

The definition of GES and GEnS in both directives requires the development of indicators, the definition of pristine or sustainable conditions and the linkage of ecological status to human pressures. The approaches in both directives related to these principles, with a focus on benthos, are highlighted in this section.

3.1. Benthic Indicators (types, comparability)

Indicators, broadly defined in the paper of (Heink and Kowarik, 2010), are a scientific response to the governmental need for reliable and accurate information on a system's conditions. For the marine environment, a wide variety of benthic indicators exist at present (Diaz et al., 2004; Borja and Dauer, 2008). The first aim of these indicators is to distinguish between a healthy and degraded water system with sufficient precision to identify the critical border between the need for 'action' and 'no action' to improve the ecological condition. The WFD and to a lesser extent the MSFD has led the implementation and fulfillment (indicators, boundaries, monitoring) of the aims of the directives by the Member states, resulting in a wide variety of assessment methods. Therefore, the WFD has to include intercalibration exercises to ensure consistency between the variety of assessment methods, used within the same eco-region and -type and to define the boundaries between the different quality classes (Borja et al., 2007). Indeed, the large scale and ambitious intercalibration exercise of the WFD promoted efficient implementation of the objective of the protection of the water systems, and greatly expanded our knowledge of indicator applications. Given that experience, future work should focus on (1) improved knowledge of natural variability within reference areas, (2) the maximization of transparency of the exercise to facilitate communication and understanding, and (3) the need for increase in the statistical power of the comparisons (Duarte, 2009). These shortcomings in intercalibration phase I can be related to major differences in available research experience between member states, to the degree of risk that each authority is prepared to accept, or to the interpretation of good ecological functioning (Mee et al., 2008). These issues are taken into account in the second intercalibration round for the WFD, which will lead to further improvement of the comparability of the indicators, used within the different Member states. In contrast, to avoid such costly (though beneficial) intercalibration the MSFD requires a common implementation of the eleven descriptors, translated into targets and indicators, at a regional sea level (Rice et al., 2010). Although defining common indicators is a step forward, the selection of the optimal and appropriate indicators still remains a major challenge. For regions and sub-regions, the optimal suite of indicators will differ for different sites, sampling effort within the different sites is unlikely to be balanced, and no single level of GEnS thresholds will be universally applicable. An increasing consistency in methods on a regional scale will probably result in the selection of more robust (widely applicable), less sensitive indicators. Intercalibration of indicators will hence be avoided in MSFD, but there will still be a need to investigate the indicators' sensitivity, to harmonize the GEnS level for any indicator and to standardize the monitoring per regional sea based on the national experience.

The benthic indicator types within the WFD include univariate, multimetric and multivariate approaches, combining in the latter different parameters with different sensitivity levels, leading to a confident assessment of the benthic ecosystem state. The indicators defined for each descriptor in the MSFD are mainly of the univariate type (abundance, biomass, productivity) and less of the multi-metric type, except for the assessment of the criterion 'the condition of benthic community' under the descriptor 'Sea-floor integrity' (Rice et al., 2010). Guidance from benthic experts on the utility of indicators tends to favor a combination of several indicators in order to evaluate the complexity of the ecosystem and to reduce the level of uncertainty of the results (Dauvin, 2007). Overall, and although useful, the use of a single univariate indicator is probably a too drastic reduction of the environmental complexity to provide a clear conclusion of the system's quality status. This has led some scientists to suggest that such indicators should be used as the basis for the computation of multi-metric or multivariate integrated indicators (Borja et al., 2004; Muxika et al. 2007a). The minimum number of ecological variables to be taken into account for multi-metric indicators should be based on studies testing the compatibility of indicators so that they do not provide conflicting information for managers or provide the same information in different ways and thus obscure overall patterns, as such avoiding redundancy (e.g. Borja et al., 2007; Gremare et al., 2009; Lavesque et al., 2009). Without these specific studies, it can be difficult to choose between different available methods (integrated, multimetric or multivariate). As such, the investigation effort in analyzing the response of univariate and multivariate benthic indicators and in testing their comparability has increased exponentially during the last decade (Diaz et al., 2004; Quintino et al., 2006; Borja et al., 2007). These studies showed that the main differences between the indicators can be attributed to (1) their differences in sensitivity (contradictory responses for the same impact), (2) their susceptibility to natural variability, (3) variable types included in multi-metric indicators (e.g. different diversity indices may react differently to the same pressure), (4) the method used for determining the sensitivity/tolerance of species, and (5) the reaction of the indicators to the sampling strategy (e.g. pooled or unpooled replicates). A benthic indicator is unlikely to be universally applicable, since organisms are not equally sensitive to all types of anthropogenic disturbance (Buhl-Mortensen et al., 2009), to geographical specifications (Dauvin, 2007) and to habitat typologies (Tagliapietra et al., 2009). An ideal indicator should be responsive to any stressor type, have a low natural variability, provide a response that can be distinguished from natural variation, and be interpretable (Hering et al., 2006).

Therefore, several indicators with complementary properties, combined in one or another way, may be needed to provide strong and effective support for management decision-making (Table 3). For the MSFD, it is necessary that the selected indicators are adequate to detect all anthropogenic impact types and that the indicators are sufficient in combination to permit an adequate global assessment (see Section 2.1). However, there is still a gap in data and knowledge in terms of measuring the function of the ecosystem using indicators, and how individual benthic animals perform their roles within specific ecosystems. Therefore, the benthic indicator types already developed in the context of the WFD should be improved to assess structural and functional benthic aspects in the MSFD, as partly proposed for the descriptor 'Sea-floor integrity'.

3.2. From pristine conditions to sustainably functioning ecosystems

The ecological status in the WFD has to be perceived or measured as a deviation from a reference condition. The identification of reference conditions/sites, with accompanying descriptions of the sites or biological elements which correspond to largely undisturbed (='near-pristine') conditions (no or minor impact from human activities), is of paramount importance for the WFD. The MSFD implementation process is not so prescriptive in this respect, because its target is sustainably functioning marine ecosystems. In this case, there is a need to set thresholds for GEnS for the different qualitative descriptors. In this context, a related problem that arises for both directives is how to define sustainable (MSFD) and reference (WFD). The WFD identifies four approaches for determining reference conditions, preferably the use of existing undisturbed sites and otherwise using historical data, models, and/or expert judgment. In practice, mainly descriptive approaches were used for determining reference conditions (Muxika et al., 2007a; Rees et al., 2008; Suding and Hobbs, 2009). Currently, no methodologies on how to define GEnS thresholds or sustainability are currently defined in the MSFD directive. Moreover, no single threshold level for any indicator will be universally appropriate within a region or subregion, due to natural geographical differences in benthic community characteristics (e.g. diversity), even within the same habitat type. This issue raises the question of how the comparability of these threshold values can be ensured between regions and subregions for the MSFD. Therefore, it is advisable to use the expert knowledge and literature obtained from the WFD process to define sustainability thresholds for the indicators of each descriptor.

Scientists are faced with the fact that there are virtually no undisturbed sites along the European coasts and estuaries, and historical data are not easily accessible, so quality assurance by experts is required (Borja et al., 2004; Tett et al., 2007). It is difficult to define how far back the baseline has to be set, which is a societal decision and depends on the historic knowledge and historical data availability of the system. In this case, experts also tend to set their own reference state or baseline employing the information from the period they felt to be "the best" (Pauly,1995; Mee et al., 2008). In fact, the lack of appropriate reference sites or robust historical datasets is one of the major problems addressed in intercalibration exercises and in setting the GES boundaries (Borja, 2005; Borja et al., 2007). This is most evident in the case of setting references in marine and coastal systems which are naturally less diverse and naturally populated with opportunistic and stress-tolerant organisms (Blanchet et al., 2008; Puente and Diaz, 2008; Puente et al., 2008; Buhl-Mortensen et al., 2009; Dauvin and Ruellet, 2009). It can be concluded that (near-) pristine sites and historical data could be the optimal ways for defining references, but they are not easily applicable for benthic communities along the European coasts and estuaries, except in Northern countries.

A recurring aspect in the above approaches is the use of expert judgment in one or another way. The WFD guidelines suggest that expert judgment may be used as a first rather than last resort, which means that member states preferably has to define their reference via one of the other approaches (undisturbed sites, historical data, modeling). Nevertheless, expert judgment may have some advantages as a complement to assessing GES and GEnS, e.g. due to the fact that benthic experts are able to reliably predict the ecological status of benthic samples, based only on species composition (Weisberg et al., 2008; Teixeira et al., 2010). Furthermore, expert judgment can also be used as an unbiased way to help and support a responsive indicator in an environmental management context (e.g., dredged material relocation and aggregate extraction), which is useful to inform scientists and regulators on wider indicator usage (Ware et al., 2010). Therefore, it is useful to apply expert judgment alongside more objective approaches.

A more objective approach should be possible in the presence of a strong stressor-response relationship with quantifiable thresholds (Borja and Dauer, 2008; Borja et al., 2009a; Josefson et al., 2009; Magni et al., 2009) (Table 3). Major inflection points derived from non-linear regressions, where benthic community structure deteriorates, could be successfully and consistently identified between different indicators and impact factors (e.g. organic enrichment, hypoxia, heavy metal pollution, physical disturbances) (Hyland et al., 2005; Zettler et al., 2007; Buhl-Mortensen et al., 2009; Fleischer and Zettler,2009;Josefson et al., 2009; Magni et al., 2009). These investigations allow determining the current status of potentially impacted areas relative to some optimal environmental quality target with more confidence (Muxika et al., 2007a). This approach is more easily applicable in systems with a clear, main pressure, e.g. organic pollution or heavy metal contamination, rather than in multi-pressure environments (Dauvin and Ruellet, 2009).

It is also very important to recognize that benthic communities change in space and over time in response to natural and anthropogenic influences (Clarke et al., 2006), possibly strengthened by climate change (Kröncke et al., 2001; Frid et al., 2009; Birchenough et al., in prep). It is therefore necessary to define 'naturalness' in the system according to best estimates of the likely boundaries and trajectories of variation and not in terms of a static pristine state (Table 3). Both 'pristine state' and 'naturalness' are therefore difficult to define (Derous et al., 2007), and are best viewed as dynamic attributes, which may need to be periodically re-defined in response to new and better scientific knowledge. Some 'pristine' targets may not be achievable for practical reasons, e.g., due to the non-linear response of many systems to measures, inadequate time and spatial scaling of the measures and differences of opinion among key stakeholders regarding the cost-benefit of implementing measures (Mee et al., 2008). The implementation of Marine Protected Area's (MPAs) could provide a solution to this problem in the future.

Defining the sustainability threshold for each indicator within regions and sub-regions will be a challenge during the implementation process of the MSFD. It will need to be based on clear stressor-response relationships, and a knowledge of the 'naturalness' of the system; expert judgment may also have a role to play (Table 3).

3.3. Pressure response of benthic communities

3.3.1. Anthropogenic pressure types

Moving from inshore to offshore, from the target areas of the WFD towards the main MSFD areas, the most influential pressure types typically change from acute eutrophication and point-source inputs of pollution to fisheries and chronic effects of dispersed inputs. Anthropogenic pressures on marine systems can be classified into two main types: large scale and often indirect impacts (e.g. climate change, eutrophication), that are not easily spatially quantifiable, and directly measurable impacts such as commercial beam trawling at the sea bed, aggregate extraction, dredging or construction, which can be more exactly located, if the existing data are made available for the assessment (Birchenough et al., 2006, 2010; Birchenough and Frid, 2009). For the first type the detection of effects relies mostly on the sensitivity of the selected indicators; a clear distinction between anthropogenic effects and climatic influences is a challenge, but examples exist (Rees et al., 2006). If the pressure intensity is spatially measurable and areas of different pressure levels or gradients can be located, a distinction may be possible simply by an appropriate monitoring design.

Furthermore, the indicators used for determining whether GES and GEnS of a given area have reached the level of "Good", need to be sensitive to these anthropogenic impact types. Both directives require the selection of indicators that are the most suited to detect the pressure type on an area-specific basis. Most benthic indicators developed for the assessment of ecological quality (Grall and Glemarec,1997; Weisberg et al.,1997; Borja et al., 2000; Simboura and Zenetos, 2002; Rosenberg et al., 2004) have been based mainly on the model of Pearson and Rosenberg (1978). This model states that benthic communities along a gradient of increasing disturbance (primarily organic enrichment) change in diversity, abundance and species composition according to their tolerance to the disturbance. This benthic response model, with differences in amplitude, is probably applicable to more pressure types, though the sensitivity or tolerance of some benthic species to certain pressure types can vary (Gremare et al., 2009). The effects of a variety of anthropogenic pressures, mainly organic enrichment related, on the performance of indicators have been extensively tested and described (Van Dolah et al., 1999; Borja et al., 2003; Muxika et al., 2005; Simboura et al., 2007; Buhl-Mortensen et al., 2009) and research is ongoing.

The main problem in evaluating indicator utility and specificity is the lack of uni-pressure response data in many systems, due to the fact that most data were collected in multi-pressure environments. This makes it very difficult or impossible to distinguish the pressure(s) mainly responsible for the benthic community changes. Another problem is related to pressures acting on very large scales (e.g., eutrophication, climate change) and where no impact-free areas exist. If measurable gradients exist (i.e. spatial and temporal) (Borja et al., 2009a; Josefson et al., 2009), such as the quantification of fishing intensities using vessel monitoring system (VMS) data, these can be used to distinguish their effects from other pressures. However, as long as these cover only gradually differing intensities, an extrapolation to undisturbed conditions is not feasible. Possible future solutions for this are the opportunities offered by MPAs that are fully protected from local-scale human disturbance but nevertheless at a scale that is appropriate for studying the different eco-system components (Table 3). More accurate quantification of certain seabed pressures is also needed (e.g. via VMS data, 'black box' data on sand extraction sites), to improve marine ecosystem assessments using indicators and to investigate pressure-response relationships (Table 3).

3.3.2. Measuring anthropogenic versus natural impacts

Indicators for ecological quality assessment in both directives need to be able to detect anthropogenic impacts and, ideally, those selected would be insensitive to natural variability. However, in practice both univariate and multi-metric indicators respond to man-induced and natural disturbance, albeit sometimes in different ways (Wilson and Jeffrey, 1994; Elliott and Quintino, 2007; Dauvin, 2007). Coastal and transitional areas are simultaneously influenced by strong natural fluctuations and disturbance events or environmental gradients such as seasonal changes (cold winters, warm summers) and salinity gradients (Reiss and Kroncke, 2005; Zettler et al., 2007) as well as by a variety of anthropogenic activities. Furthermore, anthropogenic disturbances such as eutrophication or climate change are less localized and may result in a range of direct or indirect impacts, which makes it even more important to assess the role of natural processes for indicator performance (Borja and Tunberg, 2010).

This makes an appropriate monitoring design indispensable, as often this provides the only mean to distinguish between various simultaneous influences (Table 3). An adjusted spatial pattern of monitoring stations can allow the discrimination of local effects related to locally quantifiable influences and large scale effects acting at all stations in a similar way. A comparable approach to the spatial distribution of stations is especially important, as differences in assessment procedures may yield significantly different results, even if the same indicators are employed.

Reference conditions are only partially sufficient to account for the natural variability in a marine ecosystem. These reference conditions are usually set to represent a target ecological quality for a specific habitat or eco-region. Management response is required as soon as the quality drops below this threshold value, but if the low ecological quality is caused by natural disturbances or variability, management response does not make any sense. Therefore, the evaluation of the natural 'background' variability ('naturalness') and the corresponding response of indicators are an essential requirement before establishing quality assessment strategies. However, most of the benthic indicators have been designed and used to differentiate anthropogenically impacted sites from undisturbed reference sites. Consequently, the stressor-response reaction of indicators has been extensively tested and described, whereas the information on the natural variability of indicators is very meagre (Chainho et al., 2007; Dauvin and Ruellet, 2009; Kröncke and Reiss, 2010).

Benthic data on different temporal and spatial scales have to be used to study the response of indicators to natural variability. In offshore waters these data are even less frequently available than for coastal waters. Secondly, it is of primary importance to include the 'naturalness' of the system in the reference settings. An integration of all available spatial information on pressure intensities together with a spatially well designed monitoring system will enable a more informed judgement about the differentiation between natural and anthropogenic influences (Table 3).

3.3.3. Alien species or non-indigenous species

Different definitions exist for alien (WFD) or non-indigenous (MSFD) species, but most are based on the definition as given in the Convention on Biological Diversity (CBD), i.e. 'A species, subspecies or lower taxon, introduced outside its natural past or present distribution; including any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce'. Alien species are not specifically mentioned in the WFD, but they are considered as a biological pressure. Considering their incorporation into the WFD implementation (Vandekerkhove and Cardoso, 2010), four options were proposed, of which the first two include the modification of the quality status determined by the classic classification methods, based on the presence of certain alien species. The third one assumed that the classic methods are able to detect alien species effects, and the last one proposed to use a separate 'risk assessment' for alien species based on biopollution indices (Olenin et al., 2007). For the MSFD, non-indigenous species are considered as a descriptor in the following way: 'Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems'. The indicators proposed for this MSFD descriptor focus on prevention by making inventories of non-indigenous species, target lists of potentially harmful species and the use of biopollution indices.

The duality related to this topic lies in the fact that, following the text and the spirit of the WFD (i.e. target: pristine conditions), alien species may not be present at high ecological status (Muxika et al., 2007a), whereas the MSFD requires that non-indigenous species do not adversely alter the ecosystem (i.e. target: sustainably functioning). At present, many alien species have became part of the ecosystem, mainly resulting in negative impacts (e.g. competition with and replacement of natives, habitat alteration, shifts in ecosystem functioning) (Parker et al., 1999; Nehring, 2006; Occhipinti-Ambrogi, 2007). In some cases, the invaders play a beneficial role in the ecosystem functioning (e.g. increase in biomass, filtration capacity) (Armonies and Reise, 1998; Daunys et al., 2006) or the productivity of commercial resources (e.g. aquaculture, fisheries) (Occhipinti-Ambrogi, 2007). Coastal waters are heavily exposed to introductions of alien species as a result of the high intensity of human activities (transplantations) in those regions (Reise et al., 2006; Kerckhof et al., 2007). Most of the newcomers feel very much at home in environments that are created or heavily influenced by man and through it greatly impoverished, such as harbours (marinas) and coastal areas (Reise et al., 2006; Ruiz et al., 2009). Such areas are therefore highly suitable for relatively undemanding immigrants which can outcompete the indigenous flora and fauna. This is not a local problem; there is a worldwide risk that marine flora and fauna become more standardised and that regional differences become blurred. Therefore, even if introduced species may locally increase the biodiversity, they may provoke the impoverishment of biodiversity on a larger, even worldwide scale (Occhipinti-Ambrogi, 2007).

Furthermore, in most cases, it is (almost) impossible to remove them from the ecosystem, because they are fully integrated. In such cases, experts have to evaluate the extent of integration within the natural system, and they should consider whether the effects caused by the integration of the species are adverse or not (at a functional level). When alien or non-indigenous species have become dominant, widespread and pernicious, they are termed 'invasive'. A species' tendency to become 'invasive' depends on many factors, such as its capacity for survival, reproduction and dispersal, and environmental and community suitability (Occhipinti-Ambrogi, 2007). Determining the 'invasiveness' of species is not easy, but essential for an ecological assessment (Kolar and Lodge, 2001). The ecological effect mainly depends on its function or occupied niche, so this is where research on alien species should focus (Table 3).

However, it is nearly impossible to take measures to remove or reduce the impact on the natural system once an invasive species has established itself. Therefore, prevention of further invasions is of critical importance within the directives (Table 3), due the often high economic loss associated with invasions (Pimentel et al., 2000) and the critical effect on the ecosystems. Many international fora agree to a precautionary approach, focusing on prevention of species introductions and on a quick response to alien species, even before any impacts are detected in the biological community. Therefore, an early warning system should include detection, diagnosis, quick screening, risk assessment, identification of proper response, reporting to the competent authority and an authority response (see European Strategy on Invasive Alien Species; http://ec.europa.eu/environment/nature/invasivealien/index_en.htm). Each level should be linked to potential new alien species 'alarm' lists, surveillance, monitoring, taxonomic experts, working groups and competent authorities. This has to be based on a good knowledge of the ecosystem under study as well as a thorough expertise in taxonomy otherwise new invasions will continue to arrive, unnoticed or too late. Therefore, the scientist is not only required to monitor the state of the environment, but also to predict future changes and to find ways to mitigate or manage them.

4. Monitoring requirements for environmental assessment

Both Directives state that the Member states should develop monitoring programs for evaluating the system's health and hence to allow a confident assessment of GES and GEnS. Different types of marine monitoring exist, depending on the reasons or goals for it (Gray and Elliot, 2009). In benthic monitoring, the type of sampling technique (van Veen grab, box corer, diver operated equipment, frame sampling, etc.), the number of replicates (from 3 to 20), the sample handling (sieve mesh differences) and the sampling strategy (e.g. random or fixed) depend mainly on the habitat type (e.g. intertidal, sub-tidal), the indicator type used, the expected statistical power, the goal of the program and the available budget (Muxika et al., 2007b; Van Hoey et al., 2007; Josefson et al., 2009; Lavesque et al., 2009). A difference in sampling strategy (e.g. fixed or random) has its consequences on the statistical power and the variance in the obtained data (Van der Meer, 1997). So, it is obvious that the assessment of biodiversity and its abundance (the main terms of all indicators) have different starting points in relation to the sampling strategy. Currently, the national and regional monitoring approaches vary within Europe and some aspects need to be harmonized in the light of the MSFD.

First, the scale and habitat heterogeneity of the region, area or water body needs to be evaluated. For benthic animals, it is recommended to use the habitat approach and a stratified sampling strategy, because the benthic community characteristics are habitat dependent (Van Hoey et al., 2004). However, this requires comprehensive habitat mapping, currently in development, with a good knowledge about the natural spatial and temporal variability in benthic characteristics (Degraer et al., 2008). In relation to scale, the "box strategy" could be a feasible approach, for which the directive advises to build sub-divisions within the eco-regions, related to the pressures. For this strategy, a representative choice of sampled boxes/water bodies and their habitats has to be made, which should be sampled in a repetitive way. These spatial sampling strategies need to incorporate spatially definable pressure gradients to allow a differentiation of effects from various influences and thus to allow an assessment of effectiveness of management measures.

A second aspect is the confidence that can be placed in assessments of GES and GEnS, which depends on where and how many samples were taken within a habitat, water body, sub-region or region. The number of samples that have to be taken within a habitat or water body to get a confident assessment depends on the natural heterogeneity of the habitat type and the required statistical power for detecting certain effect sizes (the level of change). The statistical power of an assessment of a habitat or water body will increase with increasing sampling effort, effect size, and will decline with increasing sample variance (Thomas and Krebs, 1997; Carey and Keough, 2002). During the development of a monitoring program, it is advisable to carefully consider these relations to reduce the error of misclassification (Underwood and Chapman, 2003).

Third, both directives require a determination of GES and GEnS every six years, thereby including the monitoring of all elements, indicators and parameters. Within the WFD, the minimum monitoring frequencies for all elements for surveillance monitoring are defined, but they are not adequate and not realistic for coastal and transitional waters, due to the high natural variability and heterogeneity within these systems (Ferreira et al., 2007). Therefore, it is better to determine the frequency of monitoring based on the variability in time of each monitored element. For example, the geomorphology of the system will be less variable in time, whereas benthic animals show large seasonal and year-to-year variations in their characteristics. Consequently, it is advisable to yearly monitor benthic animals, in a fixed period in the year (avoiding recruitment periods), with sufficient samples to confidently assess GES and GEnS after 6 years or shorter. Parameters linked to the geomorphology of the system (e.g. depth), on the other hand, needs to be monitored less frequently (e.g. once every 6 years).

Fourth, it is necessary that the monitoring is subject to standardized guidelines to ensure the comparability across regions and conclusions are not biased by inaccurate data. A standard for benthic sampling exists (cf ISO 16,665:2005; 'Water quality – Guidelines for quantitative sampling and sampling processing of marine soft-bottom macrofauna') to standardize the national programs. Aspects that need agreement on are the use of the sampling gear type, sample surface, number of samples, sampling strategy and sample handling (sieve size) to standardize the evaluation of certain indicators on regional sea levels (e.g. biodiversity). In this case, reliable and accurate taxonomy is of high importance and should be tested between participating laboratories in order to be consistent in resolution. Data quality control is a highly important step prior to index calculation, despite its time-consuming nature.

Finally, the national monitoring programs are mostly restricted by available budgets. Therefore, an objective scientific revision of the required sampling effort is needed to get an optimized, cost-effective design. Priorities have to be set within the monitoring program in relation to its objectives, the pressures present in the system, the susceptibility of the system to changes and the degree of conformity with GES and GEnS of the system (Ferreira et al., 2007). The finances can also be optimized when a horizontal approach to monitoring is taken into account, allowing resource optimization in meeting the requirements of multiple directives.

It can be concluded that both Directives, which are acting on a large scale, require the assessment of several indicators, on a regular time schedule and with sufficient sampling/analytical effort to allow assessments to be made at specified levels of confidence in the data. To successfully attain these requirements, the national and/or regional monitoring programs need to be adapted towards cost-effective monitoring strategies, which integrate the water quality monitoring, biological monitoring and supporting variables (e.g. hydrodynamics, physical parameters) (Martins et al., 2009) (Table 3). Furthermore, the monitoring locations within a system need to be flexible and the sampling density needs to be appropriate for the heterogeneity of the system. Therefore, a switch from 'station oriented monitoring' towards 'basin or system oriented monitoring', sometimes in combination with specific 'cause-effect' studies, is therefore necessary (de Jonge et al., 2006; de Jonge, 2007; Ferreira et al., 2007). Additionally, national and international monitoring programs need to be well funded and consistent.

5. Conclusion

The implementation of the WFD has led to dedicated scientific research in support of ecological or environmental assessments, with the development of several indicators, and discussions concerning the fulfillment of the principles of the directive. This activity promoted a good understanding of the strengths and weaknesses of the proposed principles in the WFD, which were widely discussed in the literature. Especially for benthic invertebrates, the accumulated knowledge is extensive, largely reflecting a long-standing preoccupation in benthic research with approaches to effective environmental assessment. The MSFD defined similar goals to the WFD, partly to avoid a need to develop new methods, but some principles are distinctly different, as identified in this contribution.

Due to the spatial extent of European marine areas and ecosystem complexity, the scope for identifying universal bio-indicators is limited, and for some, significant sampling/analytical effort is required to make a confident assessment. The implementation of well founded sampling strategies related to habitat types as well as spatially definable pressure gradients is an indispensable prerequisite for a reliable status assessment and for an evaluation of the effectiveness of management activities. Indicators deliver evidence-based information, but there are shortcomings and caution is always required concerning their use in ecological or environmental assessment. Therefore, experts have to be involved in all stages of ecological or environmental assessments at the various levels of administration (regional to EU) to ensure the quality and consistency of outcomes.

All proposed approaches have advantages and disadvantages and a cost-effective package of measures is still being evolved. Discussion and testing of approaches will lead to further insights and improvements in their selection for evaluating ecological or environmental status. Many approaches are applicable to certain regions or for certain purposes, but very few (if any) have the capability to address all problems. Therefore, care has to be taken to ensure comparable assessment strategies across the regions, to allow a region-wide evaluation of environmental status employing the same principles, even if allowance has to be made for the use of different assessment tools. The degree of applicability of approaches depends on the complexity of the methodology and their versatility across regions. The WFD has initiated and accelerated scientific research on this topic, and the MSFD can profit from it. Consequently, good communication is required between those implementing the MSFD and those implementing the WFD.

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