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Overview of eutrophication indicators to assess environmental status within the European Marine Strategy Framework Directive

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A B S T R A C T

In 2009, following approval of the European Marine Strategy Framework Directive (MSFD, 2008/56/EC), the European Commission (EC) created task groups to develop guidance for eleven quality descriptors that form the basis for evaluating ecosystem function. The objective was to provide European countries with practical guidelines for implementing the MSFD, and to produce a Commission Decision that encapsulated key points of the work in a legal framework. This paper presents a review of work carried out by the eutrophication task group, and reports our main findings to the scientific community. On the basis of an operational, management-oriented definition, we discuss the main methodologies that could be used for coastal and marine eutrophication assessment. Emphasis is placed on integrated approaches that account for physico–chemical and biological components, and combine both pelagic and benthic symptoms of eutrophication, in keeping with the holistic nature of the MSFD. We highlight general features that any marine eutrophication model should possess, rather than making specific recommendations. European seas range from highly eutrophic systems such as the Baltic to nutrient-poor environments such as the Aegean Sea. From a physical perspective, marine waters range from high energy environments of the north east Atlantic to the permanent vertical stratification of the Black Sea. This review aimed to encapsulate that variability, recognizing that meaningful guidance should be flexible enough to accommodate the widely differing characteristics of European seas, and that this information is potentially relevant in marine ecosystems worldwide. Given the spatial extent of the MSFD, innovative approaches are required to allow meaningful monitoring and assessment. Consequently, substantial logistic and financial challenges will drive research in areas such as remote sensing of harmful algal blooms, *in situ* sensor development, and mathematical models. Our review takes into account related legislation, and in particular the EU Water Framework Directive (WFD – 2000/60/EC), which deals with river basins, including estuaries and a narrow coastal strip, in order to examine these issues within the framework of integrated coastal zone management.

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1. Introduction

In its original use and etymology, 'eutrophic' meant 'good nourishment', and eutrophication meant the process by which water bodies grew more productive (Thiennemann, 1918; Naumann, 1919). About 50 years ago, however, it became clear that this 'good nourishment' had considerable environmental impacts in fresh water environments such as lakes and reservoirs (e.g. Vollenweider, 1968; Rodhe, 1969; Vollenweider and Dillon, 1974; Carlson, 1977), and subsequently similar concerns arose for estuarine and coastal systems (e.g. Ketchum, 1969; Ryther and Dunstan, 1971; Bayley et al., 1978; D'Elia et al., 1986; Lohrenz et al., 1999).

These concerns resulted in political action, translated into programmes implemented by regional conventions such as the Oslo-Paris Convention for the Protection of the Northeast Atlantic (OSPAR, 2002), the Helsinki Convention (HELCOM, 2007) for the Protection of the Baltic Sea, the Barcelona convention (MEDPOL) for the Mediterranean and into legislative instruments such as the Urban Wastewater Treatment Directive (UWWTD – CEC, 1991a) and Nitrates Directive (ND, CEC, 1991b) in the European Union (EU) and the Clean Water Act (PL 92–500, 1972) and Coastal Zone Management Act (PL 92–583, 1972) in the United States (US). Other nations also consigned into law measures for assessing and protecting the aquatic environment from eutrophication (e.g. Xiao et al., 2007; Borja et al., 2008).

The arrival of legislation led to challenges to its implementation, and a need for legal agreement on definitions. Nixon (1995) proposed that eutrophication is "an increase in the rate of supply of organic matter in an ecosystem". Although this definition was appealing to the scientific community, and correctly emphasized that eutrophication is a process rather than a state, from a management perspective it leaves substantial room for interpretation in a court of law.

As a result, by the end of the 20th Century, eutrophication had acquired a scientific and legal meaning, which in Europe was enshrined in (1) several European Directives; (2) a decision by the European Court of Justice in 2004 (ECJ, 2004); and (3) OSPAR's definition (OSPAR, 1998): "*Eutrophication means the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned, and therefore refers to the undesirable effects resulting from anthropogenic enrichment by nutrients* …."

In Europe, action against eutrophication was brought about by the conventions and legislation mentioned above, which were followed over the past decade by far more comprehensive legislation: the Water Framework Directive (WFD–2000/60/EC), which addresses all surface waters and groundwater, and the Marine Strategy Framework Directive (MSFD–2008/56/EC), which establishes a framework for marine environmental policy up to the 200 nm limit of the European exclusive economic zone (EEZ: Fig. 1). Similar to this development, though for coastal waters only, additional US legislation was passed to provide additional protection to coastal water quality (e.g. Hypoxia and Harmful Algal Bloom Research and Control Act PL 108-456, 1998 and reauthorizations in 2004, pending).

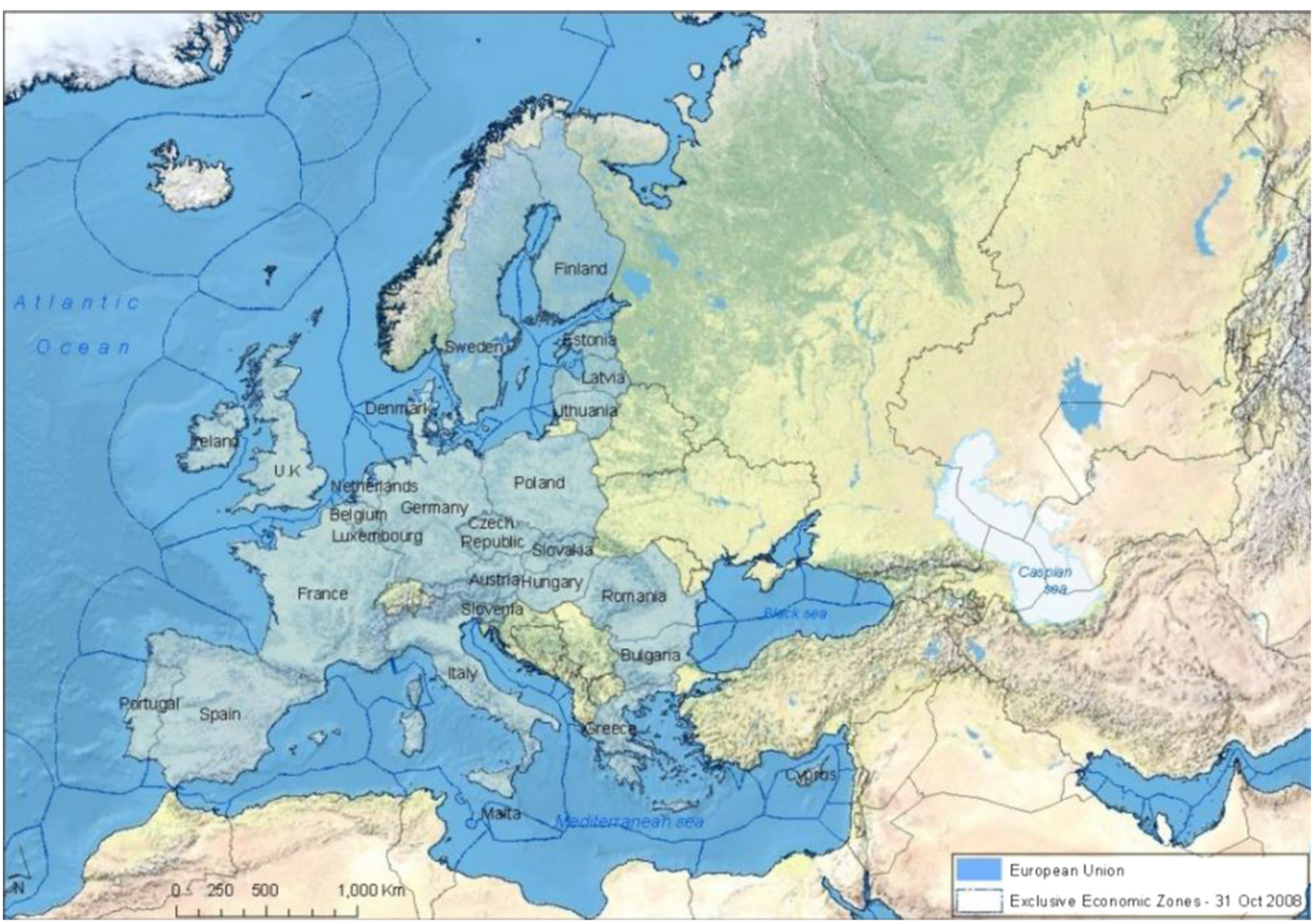

Fig. 1. Spatial scope of the Marine Strategy Framework Directive, showing maritime boundaries for EU Member States (source: JRC).

The effort that has been placed into eutrophication assessment and control in Europe over the past thirty years has resulted in: (1) systematic collection of datasets for European regional seas, in order to allow for a robust assessment of state and detection of trends; (2) development and testing of assessment methods focusing on the particular conditions that exist in marine systems; (3) construction of numerical models to relate nutrient loading, physical processes and biogeochemical cycles to state (eutrophication status), thus providing decision-makers with appropriate tools to test the outcome of management options; and (4) implementation of management measures that include the reduction of nutrient loading to coastal waters.

The WFD does not explicitly consider eutrophication (Andersen et al., 2006), and refers the word only once in Annex VIII, in the (clearly agricultural) context of nitrates and phosphates. Furthermore, because the directive adopts a "deconstructing structural" approach (Borja et al., 2010), there is no holistic eutrophication assessment model that takes into account pelagic and benthic components, since the WFD evaluates subsets of these as individual quality elements. It should be noted, however, that WFD guidance documents include assessment tools to address eutrophication.

By contrast, the MSFD takes a functional approach to eutrophication, establishing it as one of 11 holistic quality descriptors that *together* allow for environmental status assessment for European marine waters (Borja et al., 2010; Cardoso et al., 2010). The 11 descriptors are: biological diversity; non-indigenous species introduction; populations of exploited fish and shellfish; marine food webs; human-induced eutrophication; seafloor integrity; alteration of hydrographical conditions; concentrations of contaminants; contaminants in fish and other seafood; marine litter; and introduction of energy (e.g. noise). The key management objective of the MSFD is to achieve Good Environmental Status (GES) in European marine waters by the year 2020.

This contribution reports on the marine eutrophication guidance that was prepared for the EC (Ferreira et al., 2010), with the objective of informing the practical aspects of implementing the MSFD in all marine waters of the EU, and aims to contribute to the state of the art in the following areas: (1) interpretation and definition of eutrophication; (2) indicators, methods, and assessment; and (3) spatial, temporal, and policy scales, and monitoring guidelines.

2. Interpretation and definition of eutrophication

An operational definition of eutrophication was central to subsequent analysis of methodologies and scale. The approach taken was to review existing definitions in light of the MSFD, considering the following points:

1. Any definition should take into account recent developments in the scientific understanding of eutrophication, and in particular the fact that symptoms follow a well established sequence (e.g. Cloern, 2001; Bricker et al., 2003) and vary in their nature, but share a common origin: land-originated nutrient inputs. Nutrients naturally present in the sea include compounds of silicon (Si) as well as those of nitrogen (N) and phosphorus (P), in concentrations that vary seasonally, as a result of natural marine processes (Costanza, 1992; Mageau et al., 1995). Eutrophication is the result of import-driven enrichment by nutrients – primarily N and/or P – in a waterbody, which modifies the 'pristine' seasonal cycle, allowing a greater annual primary production of organic material and potentially leading to accumulation of algal biomass. The overall conceptual model for eutrophication is illustrated in Fig. 2, but it should be noted that disturbance to ecosystem compartments such as macrobenthos and fish can originate from nutrient related pressure but also from e.g. bottom trawling, overfishing, disease, etc;
2. In dealing with large marine areas, it is important to consider on the one hand the issue of spatial variability, and on the other that not all eutrophication symptoms may be relevant. For example, the loss of seagrasses (Submerged Aquatic Vegetation – SAV) is an indicator of paramount importance in the Danish Straits and German coast (Krause-Jensen et al., 2005) and parts of the Mediterranean but is inapplicable in deeper environments. Similarly, while it was felt that species shifts, and in particular those that lead to harmful algal blooms (HAB), must be an integral part of any eutrophication definition, it is important to distinguish operationally between shifts that are clearly discharge-driven, and therefore are (at least partly) amenable to management, and those that occur naturally through events such as offshore upwelling relaxation (Anderson and Garrison, 1997; Barale et al., 2008; D'Ortenzio and Ribera d'Alcalà, 2009; Siokou-Frangou et al., 2010);
3. At the scale of the MSFD, significant areas are oligotrophic, such as the Eastern Mediterranean and the northern parts of the Baltic Sea (Ignatiades, 1998, 2005; D'Ortenzio and Ribera d'Alcalà, 2009; Ignatiades et al., 2009; HELCOM, 2009). Away from the coastal fringe, nutrient related issues are different from those observed in, for example, the southern North Sea (OSPAR, 2008; Claussen et al., 2009) and Baltic Sea (HELCOM, 2009; Andersen et al., 2010; HELCOM, 2010). Since enrichment can occur naturally (Table 1), and can in some systems be an efficient stimulus e.g. to fisheries, management concern should focus on the extent to which anthropogenic nutrients may cause increases in primary production, and/or changes in N:P:Si ratios that shift the balance of primary producers from silicon-requiring diatoms towards non-siliceous algae, including cyanobacteria. These shifts may not be always be harmful, but may produce an 'undesirable disturbance' (e.g. the potential effects of increased production, and the direct and indirect changes in the balance of organisms) of ecosystem structure and function, as well as on the ecosystem goods and services used by humans (Krebs, 1988; Van De Koppel et al., 2001, 2008). However, such effects do not always result from nutrient enrichment, and may be triggered by other causes, including climate change, the removal of top predators by fishing, enrichment by allochthonous organic matter, and contamination by harmful substances. A final cause for concern is that these pressures may combine to produce larger effects (e.g. overfishing might exacerbate eutrophication problems). Thus, it is important that MSFD descriptors are not considered in isolation (Borja et al., 2010).

The MSFD eutrophication quality descriptor refers to the adverse effects of eutrophication including *"losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters"*, as described by Boesch (2002).

Oxygen deficiency can result from the sinking and decomposition of the excess organic matter produced as a result of eutrophication. It can also derive from other causes, including discharges of allochthonous organics and decreases in the ventilation of deep water caused, for example, by climate change. Ecosystem degradation is understood herein as an undesirable disturbance to the structure, vigor in function, resistance to change and resilience in recovery, of ecosystems, i.e. to ecosystem health (Tett et al., 2007; Duarte et al., 2009). Because food webs provide part of ecosystem structure, and trophic exchange contributes to ecosystem vigor, there is an overlap with the quality descriptors *concerning marine food webs* and *seafloor integrity.* Damage to ecosystem structure can include loss of biodiversity, and changes in the "balance of

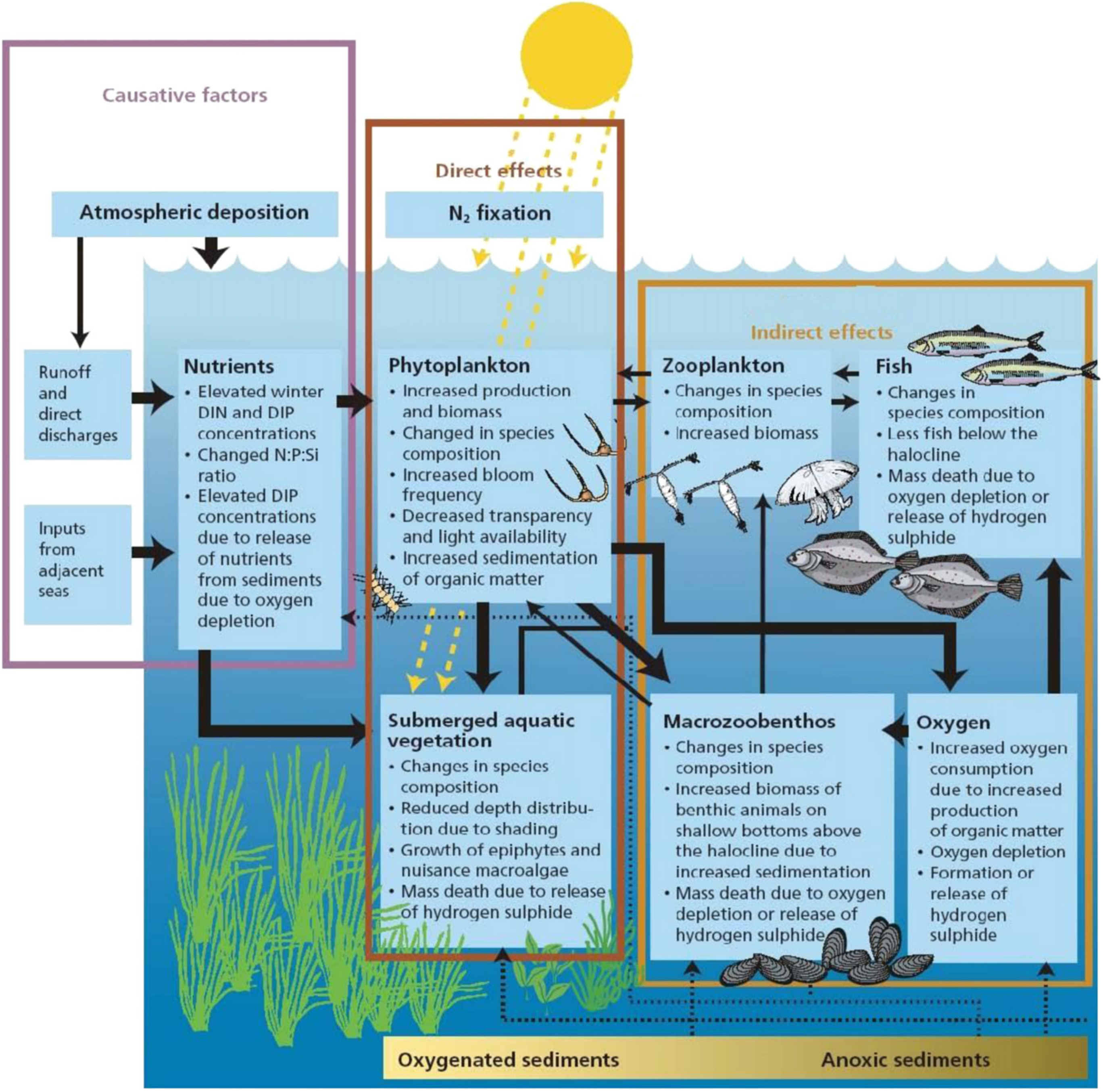

Fig. 2. Conceptual model of eutrophication. The arrows indicate the interactions between different ecological compartments. A balanced marine ecosystem is characterised by: (1) a pelagic food chain (phytoplankton ►zooplankton/zoobenthos ►fish), which effectively couples production to consumption and minimises the potential for excess decomposition (2) natural species composition of plankton and benthic organisms, and (3) if appropriate, a natural distribution of submerged aquatic vegetation. Nutrient enrichment results in changes in the structure and function of marine ecosystems, as indicated with bold lines. Dashed lines indicate the release of hydrogen sulphide (H_2S) and phosphorus, under anoxic conditions at the sediment–water interface, which is positively related to oxygen depletion. In addition, nitrogen is eliminated by denitrification in anoxic sediment. (adapted from: HELCOM 2010).

organisms" (Krause-Jensen et al., 2008; McQuatters-Gollop et al., 2009) imply a shift in relative abundances of species' populations. Thus there is an overlap with the quality descriptor *concerning biological diversity*.

Harmful algal bloom (HAB) is a broad term that embraces many phenomena (Anderson and Garrison, 1997). We will distinguish three types of harmful blooms: (1) toxic algae (e.g. *Karenia*, *Alexandrium*, *Dinophysis* and *Pseudonitzschia*) harmful to shellfish even at low algal abundance; (2) potentially toxic algae (e.g. *Pseudonitzschia*); and (3) high-biomass blooms (e.g., *Phaeocystis*, *Lepidodinium*, *Noctiluca*) that cause problems mainly because of the high-biomass itself. High-biomass blooms are sometimes called "red tides" but may in fact be brown, green or white discolourations of the sea. Some organisms (e.g. *Alexandrium*) occur in more than one category (i.e. toxic and high-biomass). Links between HABs and nutrient enrichment have been much debated. HABs should be treated as part of the undesirable consequences of eutrophication only if their frequency, amplitude, or toxicity increases in

Table 1
Definition of eutrophication, with commentary (Ferreira et al., 2010).

Definition	Commentary
Eutrophication is a process driven by enrichment of water by nutrients,	The process can be natural or human-driven, or both. Other human pressures on the marine environment can lead to similar changes and impacts, so it is a necessary condition of a diagnosis of eutrophication that the changes are linked to nutrient enrichment.
especially compounds of nitrogen and/or phosphorus,	*The main compounds are those involving nitrate, ammonium and phosphate, which are needed for algal growth; however, the decay of organic compounds of N and P can release these inorganic nutrients; and recent research has shown that organic forms such as urea can contribute directly to increased growth and may favour some harmful organisms. Attention should also be paid to changes in the ratios of nutrient -N and -P to each other and to dissolved silica, needed by diatoms*
leading to: increased growth, primary production and biomass of algae;	'Algae' is meant to refer to cyanobacterial and algal members of the phytoplankton and phytobenthos, the latter including macroalgae ('seaweeds'). We omit 'higher forms of plant life' in the present context as seagrasses can be harmed but not stimulated by the eutrophication process. We stress the centrality of 'increased primary production' to the definition, but restrict this to increased autochthonous organic production driven by increased allochthonous nutrient supply.
changes in the balance of organisms;	*Such changes are likely to take place initially in the phytoplankton and phytobenthos, and then propagate through marine food webs. The primary producer changes, which may in part result from perturbations of natural ratios of nutrient elements, include shifts from diatoms to cyanobacteria or flagellates, and the suppression of fucoid seaweeds, or sea grasses, by an overgrowth of opportunistic (green or brown) algae.*
and water quality degradation.	Such degradation includes: 'aesthetic' effects such as the appearance of red tides or excessive foam; decreases in water transparency resulting from greater biomass of phytoplankton; and decreases in bottom water or sediment pore-water oxygen content because of the decay of increased primary production
The consequences of eutrophication are undesirable if they appreciably degrade ecosystem health	*'Ecosystem health' refers to the homeostatic (self-regulatory) ability and resilience of marine food webs interacting with their non-living environment, and is evident in their 'structure' (which includes functional components of biodiversity) and 'vigour' (which includes food web function and biogeochemical cycling). Note that change in the balance of organisms is not in itself undesirable, and can occur naturally; we are concerned with nutrient-induced changes that harm ecosystem structure and function, exemplified by loss of seagrass meadows as a result of decreased water transparency, or by increased mortalities of benthic animals because of bottom water deoxygenation.*
and/or the sustainable provision of goods and services.	The nutrient-driven increase in primary production that is key to eutrophication can lead to increased harvest of fish or shellfish, as well as to undesirable consequences, such as damage to exploited fish stocks by water deoxygenation or to tourism by the accumulation of algal foam on beaches. Changes in the balance of organisms might (but don't always) include more frequent occurrences of toxic algae.

correspondence with increased nutrient input. With respect to algal toxins, there is an overlap with the MSFD quality descriptor concerning *contaminants in fish and other seafood.*

In order to account for the various aspects described above, the MSFD eutrophication guidance (Ferreira et al., 2010) agreed on the definition below as the basis for the descriptor. The steps that led to this definition, together with detailed explanations, are presented in Table 1.

Eutrophication is a process driven by enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, leading to: increased growth, primary production and biomass of algae; changes in the balance of organisms; and water quality degradation. The consequences of eutrophication are undesirable if they appreciably degrade ecosystem health and/or the sustainable provision of goods and services.

3. Indicators, methods, and assessment

Many methods have been developed in the EU and elsewhere to evaluate and track trends in eutrophication in order to fulfill requirements of legislation designed to monitor and protect coastal water bodies from degradation (see above). The progression of eutrophication symptoms is well described (Fig. 2) and most eutrophication assessment methods recognize that the immediate biological response is increased primary production reflected as increased chlorophyll *a* (Chl *a*) and/or macroalgal abundance (Bricker et al., 2007; Ferreira et al., 2007a; Xiao et al., 2007; Borja et al., 2008, 2012; OSPAR, 2008; HELCOM, 2009; Nixon, 2009; Tables 1–3). These are 'direct effects' or 'primary symptoms' and indicate the first stages of eutrophication (Fig. 2). 'Indirect effects' or 'secondary symptoms' such as low dissolved oxygen, losses of SAV, and occurrences of nuisance and toxic blooms (i.e. HAB) indicate more well developed problems (OSPAR, 2002, 2008; Bricker et al., 1999, 2003, 2007, 2008; Claussen et al., 2009; HELCOM, 2009, 2010).

Most eutrophication assessment methods integrate physico-chemical and biological indicators that provide information at an appropriate level of confidence, as a basis for management decisions (e.g. Borja et al., 2008, 2012; Zaldívar et al., 2008; Table 2). Although some methods use only selected water column parameters – i.e. Chl a, dissolved oxygen and nutrients, e.g. Trophic Index (TRIX) (Vollenweider et al., 1998) and US Environmental Protection Agency National Coastal Assessment (EPA NCA; USEPA, 2008) – others combine additional water column variables and other indicators such as the occurrence of HAB, macroalgal abundance and changes in distribution of SAV (Bricker et al., 2003). Many methods include both 'direct' and 'indirect' effects to provide the best possible evaluation of the nutrient related quality of the water body (see Borja et al., 2012; Devlin et al., 2011).

Selected indicators must show a gradient that reflects the level of human-induced impairment where an increase in nutrient loads leads to increased water quality problems. Ideally, an assessment will provide results showing the level of impairment and the concurrent load and dominant source(s) of nutrients (e.g. Table 2 and Table 3) that have caused observed impairment so that management measures can be targeted for maximum effectiveness. For example, the European Environment Agency – Environmental Monitoring and Assessment (EEA-EMMA) 'indicator comparison process' (Gelabert et al., 2008) concluded that "nutrient concentrations when used jointly with Chl a are a closer step toward a eutrophication assessment". However, nutrient concentrations may not be a useful indicator in all coastal waters.

A useful example of how the connection between loads and water quality is used for management is the Total Maximum Daily Load (TMDL) process undertaken by the US Environmental Protection Agency (USEPA, 1991). When nutrient related coastal water quality does not meet established standards (e.g. for dissolved oxygen, nutrient concentrations, aquatic plants) a calculation is made of the maximum load of nutrients that the waterbody can receive, including a margin of safety, and still meet water quality standards. Sources are

Table 2
Methods of eutrophication assessment, and examples of biological and physico-chemical indicators used, and integration capabilities (pressure-state, and overall; modified from Borja et al., 2012). Abbreviations explained throughout the text.

Method Name	Biological indicators	Physico-chemical indicators	Nutrient load related to impairments	Integrated final rating
TRIX[b]	Chl	DO, DIN, TP	no	yes
EPA NCA Water Quality Index[a]	Chl	Water clarity, DO, DIN, DIP	no	yes
ASSETS[e]	Chl, macroalgae, seagrass, HAB	DO	yes	yes
TWQI/LWQI[c]	Chl, macroalgae, seagrass	DO, DIN, DIP	no	yes
OSPAR COMPP[g]	Chl, macroalgae, seagrass, phytoplankton indicator species	DO, TP, TN, DIN, DIP	yes	yes
WFD[f]	phytoplankton, Chl, macroalgae, benthic invertebrates, seagrass,	DO, TP, TN, DIN, DIP, water clarity	no	yes
HEAT[d]	Chl, primary production, seagrass, benthic invertebrates, HAB, macroalgae	DIN, DIP, TN, TP, DO, water clarity	no	yes
IFREMER[h]	Chl, seagrass, macrobenthos, HAB	DO water clarity, SRP, TP, TN, DIN, sediment organic matter, sediment TN, TP	no	yes
STI[i]	Chl, Primary Production	DIN, DIP	no	no

[a] USEPA, 2005, 2008.
[b] Vollenweider *et al.*, 1998.
[c] Giordani *et al.*, 2009.
[d] HELCOM, 2009.
[e] Bricker *et al.*, 1999, 2003, 2007.
[f] Devlin, pers.Com.
[g] OSPAR, 2002, 2008.
[h] Souchu *et al.*, 2000.
[i] Ignatiades, 2005.

identified and loads are calculated based on concentration and flow and mass balance calculations, or more complex statistical and/or modelling approaches. Necessary load reductions are determined by comparing the TMDL to the total measured or modelled loads on a source-by-source basis. Additionally, critical conditions that influence the impact of loads are identified (e.g. rainfall, high/low flow, spills, etc). This results in an implementation plan for reductions that allocates the total load among various sources (point and non-point), and includes monitoring for effectiveness of the reductions (equivalent to operational monitoring the WFD).

Further research is needed in marine waters since eutrophication symptoms are often more clearly related to nutrient load, to susceptibility factors such as mixing and residence time, and to underwater light climate. Although the methods discussed here were developed for transitional and coastal waters, they should be considered a starting point for development of assessment methods for waters falling within the jurisdiction of the MSFD (Ferreira et al., 2010).

3.1. Considerations for indicator development: chlorophyll a

Although many multi-parameter assessment methods have been developed, the indicators that are combined and the specific manner of combination differ among the methods (Table 2 and 5). Chl a, used as a proxy for phytoplankton biomass, is common to all methods and there is extensive literature on its use as an indicator in inshore and offshore waters (Bricker et al., 1999, 2003, 2005, 2007, 2008; Kowalewska et al., 2004; Zaldívar et al., 2008; Borja et al., 2008, 2012; Boyer et al., 2009; Claussen et al., 2009; Garmendia et al., 2011; Carstensen and Henriksen, 2009; Devlin et al., 2007, 2009; HELCOM, 2009). Though all assessment methods include Chl a, the metrics used differ. The Chl a indicator is thus a good example of the variability that exists among indicator formulations and highlights important considerations for indicator development. For example, though the thresholds and ranges of Chl a concentrations for transitional water classification are notably similar among methods, the timeframe and spatial scales of

Table 3
Pressures and impacts to be considered for the eutrophication Quality Descriptor, as defined in Tables 1 and 2 of Annex III of the MSFD.

	Characteristics		Pressures and impacts
Physical and chemical features	Spatial and temporal distribution of nutrients (DIN, TN, DIP, TP, TOC) and oxygen, pH, pCO2 profiles or equivalent information used to measure marine acidification[a]	Nutrient and organic matter enrichment	Inputs of fertilizers and other nitrogen and phosphorus-rich substances (e.g. from point and diffuse sources, including agriculture, aquaculture, atmospheric deposition), Inputs of organic matter (e.g. sewers, mariculture, riverine inputs)
Biological features	A description of the biological communities associated with the predominant seabed and water column habitats. This would include information on the phytoplankton and zooplankton communities, including the species and seasonal and geographical variability	Nutrient and organic matter enrichment	Changes in production
	Information on angiosperms, macroalgae and invertebrate bottom fauna, including species composition, biomass and annual/seasonal variability	Nutrient and organic matter enrichment Physical alteration	Changes in production, changes in spatial coverage of bottom flora and fauna

[a] Under the slightly more alkaline conditions associated with eutrophication a reduction in pCO2 and increase in pH would be expected.

Table 4
Methods to evaluate the status of phytoplankton in coastal and estuarine water bodies (modified from Borja et al., 2012).

Method	Area using method	Biomass				Community composition	Abundance	Indicators in Overall Eutrophication Index
		Chl a thresholds and ranges (ug l^{-1})	Sample timeframe	Statistical measure	Other characteristics			
EPA NCA[a]	US	Poor > 20; Fair 5–20, Good 0–5; lower for sensitive systems	Index period (June–Oct)	concentration, % of coastal area in poor, fair and good condition based on probabilistic sampling design for 90% confidence in areal result		No		Chl a, water clarity, DO, DIP, DIN
TRIX[b]	EU	no thresholds, integrated with other index variables		concentration		No		Chl a, DO, DIN, TP
TWQI/LWQI[c]	EU	Good QV100 = 6; Bad QV0 = 30	annual	Chl concentration mean annual or seasonal modified by weighting factor		No		Chl a, seagrasses, macroalgae, DO, DIN, DIP
HEAT[d]	Baltic	Deviation from ref EQR <0.67; No dev from ref EQR >0.67	summer (June–Sept)	mean summer concentration	increases in concentration, frequency and duration	indicator spp	X	Chl a, phytoplankton, nutrients, water transparency, SAV, DO, benthic invertebrates, summertime bloom intensity index
ASSETS[e]	US, EU, Asia, Australia	High >20; Mod 5-20; Low 0-5; lower for sensitive systems	annual	90th percentile Chl concentration of annual data	spatial coverage, frequency occurrence	Nuisance and toxic bloom occurrence, frequency, duration		Chl a, macroalgae, DO, seagrasses, nuisance/toxic blooms
WFD[f]	Basque Country	Cantabrian coast: Bad >14, Poor 10.5–14, Moderate 7–10.5, Good 3.5–7, High 0–3.5	summer	summer Chl concentration mean, max and sometimes 90th percentile annual data	increases in concentration, frequency and duration	indicator spp	X	Chl a, phytoplankton, macroalgae, microphytobenthos, seagrasses, DO, nutrients, algal toxins
WFD[g]	UK	Mediterranean coast ($P90^{th}$): T2 (34.5 < sal <37.5) A: H/*G* = 2.4 (EQR = 0.80); *G*/M 3.6 (EQR = 0.53) T3 (sal>37.5) W-Med: H/*G* = 1.1 (EQR = 0.80); *G*/M 1.8 (EQR = 0.50). E-Med: H/*G* = 0.1 (EQR = 0.80), *G*/M 0.4 (EQR = 0.20)	At least 5 years data available, with monthly sampling, in the surface layer	EQR based on Chl concentration mean or 90th percentile	Mean salinity or density	No	No	Biological quality elements (phytoplankton, macroalgae, macroinvertebrates, seagrasses)
OSPAR COMPP[h]	North East Atlantic	NPA if below RC+50%, PA if above RC+50%	growing season	growing season Chl concentration mean, max	increases in concentration, frequency and duration	indicator spp	X	Chl a, phytoplankton, macroalgae, microphytobenthos, seagrasses, DO, nutrients, algal toxins
IFREMER[i] (lagoons)	France	> 30 Red; 10-30 Orange; 7-10 Yellow; 5-7 Green; 0-5 Blue	annual	mean annual Chl concentration		phytoplankton abundance of <2 μm, >2 μm	X	Chl a, phytoplankton counts (<2, >2 μm), macrophytes (biomass, diversity), macrobenthos (richness, diversity), water (DO, Chl, Chl/phaeo, turbidity, SRP, TP, TN, NO_2, NO_3, NH_4), sediment (OM, TN, TP)

[a] EPA (Environment Protection Agency) (USEPA, 2005, 2008).
[b] Vollenweider et al., 1998.
[c] TWQI/LWQI (Transitional Water Quality Index) Giordani et al., 2009.
[d] HELCOM, 2009.
[e] Bricker et al., 2003, 2007.
[f] WFD (Water Framework Directive) Devlin et al., 2009.
[g] European Commission, 2008.
[h] OSPAR COMPP (OSPAR Comprehensive Procedure) OSPAR, 2002, 2008.
[i] Souchu et al., 2000.

Table 5
Tentative list of eutrophication indicators and timeframes for marine waters assuming samples are taken on a spatially representative basis (see Table 4 for alternative approaches).

Indicator Type	Indicator	Sampling timeframe[a]	Statistics
Pressure	Nutrient load (Nitrogen, Phosphorus)	Annual estimate to match timeframe of eutrophication condition assessment	Tons/year can be calculated from riverine and direct inputs adjusted to the inflow, industrial and urban water treatment plant loads. OSPAR RID Programme and HELCOM Pollution Load Compilations (PLCs) could be used for guidance.
State or Condition	Increase in primary production	Estimates at some periodicity over the annual cycle	Can use chlorophyll and other algal components as a proxy or use remote sensing plus modelling as appropriate and as resources allow
	Chlorophyll	Monthly, or more frequent as appropriate and as possible especially for dynamic areas	90^{th} percentile concentration, spatial area of high concentrations
	Dissolved oxygen	Monthly, or more frequent as appropriate and as possible especially for dynamic areas	10^{th} percentile concentration, spatial area of low concentrations
	Opportunistic macroalgae	Annual sampling in spring – summer when blooms are more probable	Blooms that cause detriment to living resources, duration of blooms, approximate spatial coverage of blooms
	Nuisance/toxic algal blooms	Annual bloom events Annual to multi-year changes in frequency and/or duration of blooms	Blooms that cause detriment to living resources
	Changes in algal community structure	Annual to multi-year changes from fucoids/kelp to opportunistic green/brown algae and/or changes in balance of diatoms/flagellates/ cyanobacteria	Change from diverse natural community to one dominated by opportunistic and/or nuisance and/or toxic species
	Submerged Aquatic Vegetation	Annual surveys	Changes in: spatial coverage, density of beds
	Benthos	Annual	Changes in diversity and proportion of sensitive versus non-sensitive species
	Nutrient concentrations	Monthly or fortnightly, or more frequent as appropriate and as possible especially for dynamic areas	Annual means or maxima, Seasonal means or maxima, others as appropriate
Other	Benthos/fish	Observations/irregular – take note of kills	Massive mortality, benthos/fish kills

[a] More frequent sampling on a temporal basis and more samples spatially for better areal representation may be appropriate and justified (e.g. surveillance monitoring of WFD), particularly for problem areas and those at risk, but it must be balanced with consideration of resources available for monitoring.

sampling, the statistical measures used to determine representative concentrations (e.g. mean annual, index period mean and/or maximum, 90th percentile; Table 4), the determination of reference conditions and the combination of characteristics for the final status rating are different.

4. Statistical measures, determination of reference conditions, and indicator formulations

Equally important to the timing and spatial representativeness of samples are the statistical measures used to determine indicator concentrations, the determination of reference conditions that represent the acceptable/desired concentration, and the formulation of the indicator. Again using Chl a as an example, in the USEPA NCA (USEPA 2001, 2005, 2008) and ASSETS (Assessment of Estuarine Trophic Status) (Bricker et al., 1999, 2003) methods, reference conditions and concentration ranges are determined from national studies. While they are relevant for most estuaries, some adjustments (i.e. different scaling) are made for more or less sensitive systems; areas within the MSFD framework will likely need similar types of adjustments. The EPA NCA method uses measured concentrations compared to Reference Conditions (RC) to determine the rating for each sample station and a ratio of good/fair to poor/missing from all sampling stations to determine the final rating. The ASSETS method uses the 90th percentile of annual data compared to RC. The ASSETS method includes the spatial coverage of high concentrations, and the frequency of occurrence of blooms in the formulation to provide a comprehensive picture of Chl a condition.

The IFREMER (Souchu et al., 2000) method uses the 90th percentile of annual or seasonal Chl a data which is compared to a fixed scale RC determined from studies such as those of the Organization for Economic Cooperation and Development (OECD; Vollenweider, 1968) which are consistent with the scales reported for EPA NCA and ASSETS (Table 4).

The Transitional Water Quality Index (TWQI/LWQI) (Giordani et al., 2009) method uses non-linear functions to transform annual average Chl a concentrations from sites representative of the system into a Quality Value (QV) that is then multiplied by a weighting factor that accounts for the relative contribution of Chl a to the overall index. The Chl a QV scores, are consistent with the reference condition scales of the EPA NCA, ASSETS and IFREMER.

The HELCOM Eutrophication Assessment Tool (HEAT) method (Andersen et al., 2010 and HELCOM, 2009), the OSPAR COMPP (Topcu et al., 2009) and WFD determine RCs from historical data, empirical modelling or ecological modelling for pristine conditions. Historical data and modelling are especially valuable in systems governed by internal loads and switches in redox conditions; under such conditions "pure statistics", e.g. trend analysis, may be misleading.

The HEAT method and WFD determined methods use an Ecological Quality Ratio (EQR) approach while for the OSPAR COMPP (Claussen et al., 2009), a Problem Area is indicated if measured Chl a is greater than the RC+50%. The WFD RCs were developed during intercalibration exercises and reflect the location of the assessment, e.g. Basque coast (European Commission, 2008; Revilla et al., 2009). The WFD assessments use both 90th percentile and the mean of Chl a for the vegetative growth period as indicators of phytoplankton biomass (Table 4). The Statistical Trophic Index (STI; Ignatiades, 2005) assesses the trophic status of sea water using seasonal data for Chl a and primary production. The data are scaled statistically through the analysis of probabilistic parameters. This analysis estimates the limits of average concentrations in the relationship eutrophic > mesotrophic > oligotrophic for Chl a, primary production, and physico-chemical parameters by defining thresholds and reference conditions among inshore, offshore, and open ocean waters. Unlike the other methods, the TRIX method does not use reference conditions or scaling for Chl a individually, having only a scale for an integrated rating with four other indicators (Table 4).

These existing methods provide guidance about important considerations for inclusion in indicator development. While Chl a is used here as an example, the same framework with respect to the spatial and temporal sampling and use of indicator characteristics (e.g. concentration, spatial coverage, frequency of occurrence) should be considered in developing other biological and physico-chemical indicators. These methods (Table 2 and Table 4) should also be used to determine how to combine indicators into a comprehensive multi-parameter assessment of eutrophication.

4.1. *Confidence evaluation*

Finally, the methods that are developed should include an evaluation of the confidence for each indicator and for the overall eutrophication status rating. Given the different spatial scales and timeframes of data that might be used and compared among different water bodies, as well as the different ways to develop RCs, it is highly recommended that the results have an associated level of confidence. At present there are two methods to consider for development of this type of assessment. Bricker et al. (1999, 2003, 2007) use the availability and confidence (based on spatial coverage and analytical considerations) of data to determine a Data Confidence and Reliability assessment. The evaluation developed by Andersen et al. (2010) includes a combined evaluation of confidence in RCs, deviation from RCs and the actual status of the water body. These methods are strongly dependent on expert knowledge but they are useful as a starting point for development of an evidence-based confidence rating to accompany the eutrophication status rating in marine waters. This is particularly important given the likelihood that assessment methods will be developed differently to address conditions within specific regions.

4.2. *Recommended indicators for monitoring and assessment*

The eutrophication indicators that should be monitored in marine waters can be derived from previous studies (Table 5), though there may be others that are more relevant and SAV may not be appropriate in deeper waters.

To provide a complete picture of eutrophic conditions, other characteristics in addition to Chl a should be included, such as changes in community composition, occurrence of nuisance and potentially toxic species that result from changes in nutrient ratios, and increased duration and frequency of blooms that result from increases in nutrient loads (Table 5).

Most pressures resulting in eutrophication come from coastal areas, producing a strong gradient from coastal to offshore waters; consequently it is recommended that the WFD assess the status in coastal waters using all elements (biological and physico-chemical) affected by eutrophication (Table 4). This must then be complemented, within the MSFD, using phytoplankton assemblage and physico-chemical (e.g. nutrients, transparency, etc.) indicators in offshore and open marine waters (Borja et al., 2010).

It is fundamental to include nutrient sources and loads (e.g. terrestrial, airborne) in the overall assessment so that loads can be associated with impairment and successful management measures can be developed from that relationship (Bricker et al., 2007; OSPAR, 2008, HELCOM, 2009). The US EPA TMDL process (see section 3; http://water.epa.gov/lawsregs/lawsguidance/cwa/tmdl/) might be considered as a starting point given it was developed for coastal waters. Another possible tool is the Indicator of Coastal Eutrophication Potential (ICEP) (Billen and Garnier, 2007), which estimates potential eutrophication impacts from riverine nutrient loads on the basis of their N:P:Si ratios. The framework for a monitoring program should also be guided by established assessment procedures, such as the OSPAR Comprehensive Procedure (OSPAR, 2002, 2008). For example, to maximize the efficiency of monitoring as well as resource use, a screening process might be applied whereby only water bodies showing impairment or risk from anthropogenic nutrient loads in an initial assessment would be the focus of a more intensive monitoring and assessment program. The initial screening should be done periodically to ensure that any creeping eutrophication would be detected.

5. Spatial, temporal, and policy scales, and monitoring guidelines

5.1. *Spatial scale*

5.1.1. *Effects of increasing the nutrient load*

Eutrophic areas are primarily located near the coast (e.g. Diaz and Rosenberg, 2008), because nutrient enrichment due to land based inputs to coastal waters is the first factor promoting eutrophication. Although these are typically sensitive areas receiving anthropogenic nutrient loading, some natural symptoms of eutrophication can also be found in upwelling regions, sedimentation areas, or frontal systems. An increase in nutrient discharge to coastal areas could lead to increased phytoplankton biomass during the spring bloom, but also to the emergence of additional episodic blooms during summer and autumn (e.g. Cugier et al., 2005). For European seas, satellite maps compiled from summer data show a very heterogeneous distribution of highly productive areas along the European shores. While the whole shallow south and eastern North Sea, a significant part of the Baltic Sea, and the Black Sea, are highly productive, the Atlantic and Mediterranean exhibit only a strip of high production along the coast. It should, however, be noted that current algorithms for processing remotely sensed sea colour may overestimate chlorophyll in waters containing high levels of coloured dissolved organic matter (e.g. the Baltic) or suspended particulate matter (e.g. the North Sea).

The EUTRISK (Eutrophication Risk) index (Druon et al., 2004) and the OXYRISK (http://emis.jrc.europa.ec/) maps the risk of summertime eutrophication and oxygen deficiency in EU coastal waters. Extensive risk areas include: (1) large parts of the Baltic, including the central and southern areas; the exceptions are the northernmost region, the Kattegat and coastal water in the Skagerrak; (2) the central and southern North Sea and the coastal waters west of Jutland; (3) the Azov Sea and western coastal belt of the Black Sea; (4) the northern Adriatic Sea, and the northern French coast of the Bay of Biscay. In the case of the Baltic and northwestern EU waters, these areas largely correspond to those identified by the HELCOM thematic assessment as 'eutrophic', and by the OSPAR comprehensive procedure as 'Problem Areas'.

5.1.2. *The role of bathymetry and hydrodynamics*

The risk of eutrophication is linked to the capacity of the marine environment to confine growing algae in the illuminated surface layer. The geographical extent of potentially eutrophic waters along European coasts may vary widely, depending on:

(1) extent of shallow areas, i.e. with depth $\leq$ 20 m;
(2) degree and extent of water column stratification. Stratified river plumes can create a shallow surface layer separated by a halocline from the bottom layer, whatever its depth. The potential for eutrophication is high where nutrients are introduced into the surface layers of semi-enclosed water bodies such as fjords or rias that have long periods of water column stratification due to river discharge and/or the deep intrusion of dense coastal water. The risk increases with increasing water residence time;
(3) water residence time. Long water residence times in enclosed seas leading to blooms triggered to a large degree by internal and external nutrient pools;
(4) occurrence of upwelling phenomena leading to nutrient supply and high nutrient concentrations from deep water nutrient pools, which can be of natural or human origin;
(5) occurrence of sedimentation areas or frontal systems where nutrients and organic matter concentrate due to their hydrographic characteristics.

A good example of the combination of features (1) and (2) is provided by the southern and eastern parts of the North Sea; this shallow (<50 m deep) and tidally mixed region receives, cumulatively from SW to NE, the majority of the riverine nutrient loads to the North Sea (Seine, Thames, Scheldt, Rhine, Ems, Weser, Elbe; Lancelot et al., 1987).

5.2. Temporal scale

5.2.1. Effects of changing the nutrient balance

Except in permanently stratified, deep areas, such as the central Baltic Sea, the acute quantitative symptom of eutrophication, i.e. severe hypoxia, is a seasonal feature, which occurs after strong primary production episodes, mainly in late spring or in summer, when calm weather and seasonal formation of a pycnocline prevent atmospheric oxygen from being brought to deep water layers.

At the qualitative level, eutrophication may alter the natural succession of species during the year. The terrestrial waterborne loadings on the European coastal shelf have varied during the last century in a nearly independent way for the three main nutrients N, P and silica (Si). Whereas Si remained quasi-constant or slightly declined due to partial trapping by settling fresh water diatoms upstream of dams, P increased until the 1990's, and then decreased due to the polyphosphate ban in detergents and phosphate removal in sewage treatment plants (e.g. Billen et al., 2001, 2005). Nitrogen increased continuously during the second half of the 20^{th} century, but began to slightly decrease during the last decade due to implementation of European legislation such as the Nitrates Directive (ND; CEC, 1991b) and the Urban Wastewater Treatment Directive (UWWTD – CEC 1991a). The National Emission Ceilings Directive restricts inter alia atmospheric nitrogen emissions per Member State and facilitates significant reductions of oxidised and reduced nitrogen.

The changes in N:P:Si balance have induced some shifts in the phytoplanktonic flora, both in the abundance of diatoms relative to other groups, and in the relative importance of (regional) indicator species. In the Greater North Sea, for instance, undesirable blooms of two haptophytes have been recorded. *Phaeocystis globosa*, which forms spherical colonies with foam as by-product, invades the coastal strip off France, Belgium, the Netherlands and Germany every spring (April–May; Lancelot, 1995). The toxin-producing *Chrysochromulina* spp., which blooms between April and August in the Kattegat and Skagerrak (Dahl et al., 2005), was responsible in May–June 1988 for an extensive episode of toxicity decimating fish farms. These haptophytes are known to follow the early-spring diatom bloom (Rousseau et al., 2002; Dahl et al., 2005) when a remaining excess of nitrate allows their rapid growth, even if phosphate conditions are low (Lancelot et al., 1998; Dahl et al., 2005), because both species are mixotrophs, being able to use organic forms of phosphorus (Veldhuis et al., 1991; Paasche, 2002). In the Baltic, the decrease of Si levels and concurrent increase of N and P inputs have led to a flagellate dominance in some areas and to elevated production and sedimentation. A similar situation was observed in the NW Black Sea in the mid 1970s where the nearly simultaneous increase of N and P and decrease in Si led to the dominance of *Prorocentrum cordatum* (a harmful species) over diatoms. In the Black Sea, the N:P:Si imbalance was exacerbated by Si retention in reservoirs in the Danube (Humborg et al., 1997). Presently, however, all three nutrients have decreased for different reasons allowing a trend toward a more natural balance in N:P:Si stoichiometry (Yunev et al., 2007).

Along the Atlantic and English Channel coasts, several harmful species of phytoplankton have been recorded, producing diseases in human consumers of shellfish. Some of them are dinoflagellates, and may have been triggered by summer excess nutrient in the coastal plumes (Guillaud and Ménesguen, 1998).

In the Baltic Sea, the increased magnitude and frequency of cyanobacterial blooms (including toxic species like *Nodularia spumigena*) have been related to increased nutrient concentrations (both N and P) during the last decades. Elevated nutrient inputs, maintaining increased phytoplankton spring bloom production and subsequent sedimentation, leading to an extension of anoxic bottoms and triggering regeneration of P from sediments, are part of a vicious circle where external nutrient loading (both N and P) enhances the occurrence of cyanobacterial blooms in the Baltic (Vahtera et al., 2007; HELCOM, 2009).

The coastal waters of the western Aegean Sea (E. Mediterranean) have not been prone to seasonal blooms of the invader species *Alexandrium minutum* because the local nutritional status did not support its N:P ratio requirements and the phytoplankton communities were dominated by diatoms that were strong competitors of this species (Ignatiades et al., 2007).

5.3. Policy scales

As a result of the WFD, European Member States have delineated coastal water bodies (e.g. Ferreira et al., 2006), however in most cases, since the seaward limit is defined in the directive as “a distance of one nautical mile on the seaward side from the nearest point of the baseline from which the breadth of territorial waters is measured”, such water bodies miss the largest part of wide eutrophic plumes. Turbidity in some estuarine and near-coastal waters is often too high to allow strong primary production, whereas enriched surface waters further offshore can host very productive communities when suspended inorganic particles have settled.

GES has to be set for areas within the European EEZ, based on eutrophication parameters that will be part of the monitoring programmes. Such areal delineation should be based on oceanographic characteristics, such as the Physically Sensitive Area (PSA), the JRC OXYRISK and EUTRISK indices (Druon et al., 2004; http://emis.jrc.ec.europa.eu/), and the appropriate subdivisions used by HELCOM (2009) and OSPAR.

Some improvement in these indices would probably be gained by using new techniques for revealing the dynamically confined areas in the open coastal ocean, such as remote sensing combined with numerical modelling (Ménesguen and Gohin, 2006), as well as tracking the far-field impact of national river loadings (Ménesguen and Gohin, 2006) to assess transboundary effects (e.g. OSPAR's Intersessional Correspondence Group on Ecosystem Modelling [ICG-EMO] OSPAR, 2009). Modelling may provide new insights into long-range effects which are difficult to measure by field sampling techniques. Enclosed seas such as the Baltic, where eutrophication is impacting almost the whole area, require a regional approach, where delineation of areas and related GES targets are based on evaluation of long-term development and on-going modelling work of the expected impacts of nutrient loading reductions, e.g. as planned by the Baltic Sea Action Plan (Wulff et al., 2007; HELCOM, 2009). The next step will be to set clear GES criteria for eutrophication parameters for these areas. Lessons may be learned from the Baltic Sea, where visions and goals have been agreed via the Baltic Sea Action Plan and the process of setting targets has been started, and from a similar process currently being developed by OSPAR. In the US, a parallel can be drawn for the Gulf of Mexico (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. 2008).

5.4. Monitoring guidelines

5.4.1. Spatial and temporal scales

The spatial and temporal monitoring framework is an important issue in the determination and confidence of final assessment results (Carstensen, 2007; Andersen et al., 2010). Sampling is designed to capture extreme or problematic events or time periods; ideally samples would be taken year round to observe both baseline and bloom concentrations. However, when resources are limited sampling is usually restricted, and in places with strong seasonal variability may be limited to samples from the typical bloom period to try to capture peak concentrations, usually the spring or summertime growing season (or winter sampling in the case of nutrients). In marine areas with less well defined seasonality, sampling on an annual cycle may be more appropriate despite resource considerations and in these cases, remote sensing is suggested as a potential solution to overcome these issues (Ferreira et al., 2010).

Alternatively, a sampling design could include consideration of both natural characteristics and the human dimension to divide a water body into management units where morphology as well as appropriate indicators of pressure and state would determine zone boundaries as well as sampling locations and frequencies (Ferreira et al., 2006). The benefit of this approach is that special monitoring and management can be implemented in cases where there is a particularly impacted area.

The sampling framework is addressed differently by the different assessment methods from a one time sampling per index period (i.e. EPA NCA) to monthly sampling during an annual period (Table 4). In marine regions the identification of temporal trends in Chl a concentration is important, but the sampling resolution in time (e.g. once a year for the NE Atlantic) and space (very limited station network in some regions) may make trend analysis difficult (Gelabert et al., 2008).

The spatial coverage of Monitoring Programmes designed to comply with the MSFD may be divided into (a) a coastal strip where the WFD is also enacted; and (b) a more extended marine area (Fig. 1). In the former, the combination of surveillance, operational and investigative monitoring put in place by Member States for WFD compliance (e.g. Ferreira et al., 2007b; Borja et al., 2010) is also appropriate for MSFD compliance with respect to eutrophication assessment. The design of Monitoring Programmes for open marine water must take into consideration the strong diversity of EU regional seas.

In some cases, such as the Baltic, the whole marine area is bounded by limits of territorial waters, and in others, such as the eastern Mediterranean or NE Atlantic, there are marine areas that are international waters. Nevertheless, most of the offshore areas subject to the MSFD generally show limited eutrophication symptoms (Ærtebjerg et al., 2001; Frid et al., 2003). Indirect eutrophication effects (secondary symptoms) such as hypoxia are not observed, except in the Baltic Sea (HELCOM, 2009). In the Black Sea hypoxia has been a naturally occurring oceanographic phenomenon for much longer than the time-scale of human influence on water quality (Sorokin, 2002). This is also well documented in other parts of the world, such as the Cheseapeake Bay in the US (Cooper and Brush, 1991).

Due to the wide extent of eutrophic zones in some coastal parts of the European seas, the sampling effort necessary to reliably assess algal biomass will increase significantly in some countries with respect to WFD requirements. Hence, a systematic use of remote sensing of the surface chlorophyll content and other automated sampling techniques such as buoys, ferry boxes, and gliders, are recommended, and should be regularly improved by comparison to more conventional sampling techniques. This approach, associated with the use of models, has allowed a systematic coverage in time and space of the national WFD water bodies (Gohin et al., 2008). In the case of high-biomass HAB, satellite remote sensing of Chl a will probably pick up the signal, with the caveat that when the bloom is not superficial (e.g. when present in thin deeper layers as in the English Channel), it will be a challenge for satellite detection. Toxic phytoplankton patches with low biomass, i.e. close to background concentrations, are also particularly difficult to monitor. In both cases, the development of HAB-specific algorithms is an important research recommendation.

Several US modelling efforts use satellite and field data to identify blooms as HAB or non-HAB, predict conditions favorable for occurrence of *Karenia brevis* (Stumpf et al., 2003) and *Alexandrium fundyense* (McGillicuddy et al., 2005; Li et al., 2009), characterize bloom distribution and intensity of HABs (e.g. *Microcystis aeruginosa*, Wynne et al., 2010), and predict transport of HAB blooms (McGillicuddy et al., 2005; Wynne et al., in press). One study in particular, focused on *Karenia brevis* in Florida coastal waters (Stumpf et al., 2009), is a good example of how this type of research can be developed into an operational forecast system. While these studies are focused on coastal waters, they have shown promising results and may serve as a starting point for development of HAB identification, characterization and forecasting capabilities in waters under the jurisdiction of the MSFD.

Eutrophication indices based on monitoring and/or modelling must consider temporally appropriate datasets, which may (1) favour seasonal datasets (e.g. the productive period and/or winter nutrients); or (2) an annual cycle, which may be more adequate for marine areas with less well defined seasonality. In order to detect acute effects, which often pose serious threats to the ecosystem, monitoring and modelling must be temporally adjusted to rapidly developing events, such as sudden and sharp peaks of oxygen depletion in bottom waters. This requires use of several approaches combining studies onboard research vessels with high-frequency automated sampling onboard ships-of-opportunity, satellite imagery, models, automatic high-frequency buoy recordings, and traditional sampling in marine areas that are impacted or at risk of being impacted by eutrophication. Measured data may provide ocean boundary conditions for the WFD coastal area, and help establish the cause of violation of quality thresholds for some indicators.

As in any regional (and transboundary) framework, EU Member States must determine to what extent data needs are covered by national monitoring programmes, and what aspects of the eutrophication assessment are adequately covered. Any monitoring programme must include appropriate quality assurance, allowing for appropriate intercalibration and comparative assessment, and should be guided by existing programmes, such as the OSPAR Comprehensive Procedure (OSPAR 2002, 2008, 2009) and HEAT (HELCOM 2010). Accordingly, it will be possible to optimize existing monitoring information, and identify where improvements may be made through targeted/focused additional monitoring.

5.4.2. Infrastructure improvements

A long-term monitoring and research infrastructure is needed, including marine/oceanic observation capabilities that include continuous plankton recorders and long-term fixed stations of data collection for model validation. Maintenance of long-term data series and information is important for prevention of misdiagnosis of new events/changes and will improve interpretation of trends in HAB and facilitate development of management measures.

6. Conclusions

The work carried out by this MSFD guidance task group identified a number of research areas where increased effort should be placed in order to improve assessment capabilities and thus the potential success of management measures:

6.1. Nutrient inputs

- Estimates of nutrient loads from terrestrial and atmospheric sources, in relation to transitional/coastal retention, and chemical and biological target indicators;
- Determine natural background nutrient enrichment (e.g. upwelling, import from pristine/good status rivers) compared to human related sources for determination of unimpacted state, and distinction between naturally productive status and anthropogenically eutrophic status for identification of what can and cannot be managed, the development and use of ecosystem models is necessary to assist the estimate of this contribution;
- Determine the contribution of transboundary and transnational supply and/or exchange of nutrients compared to terrestrial and atmospheric sources of nutrients and whether/how these can be managed;
- Evaluate potential climate change impacts on availability of nutrients including transportation (e.g. from new circulation patterns, increased rainfall, changes in upwelling/coastal processes that might lead to new or enhanced sources), and transformation of nutrients and organic matter;
- Determine how to distinguish between climate change and anthropogenic impacts and how best to manage these;
- Evaluate relationships between indicators/parameters and proxies for nutrient loading pressures (e.g. change in nutrient concentrations where this can be demonstrated to be an effective proxy) in order to set ecoregion and/or habitat-specific targets for GES.

6.2. Primary production and algal biomass regulation

- The relationship among nutrient concentrations, chlorophyll, and primary production, and whether when used jointly they are useful and should be pursued as part of eutrophication assessment, given the stronger linkage of symptoms to nutrient loading, underwater light climate and susceptibility (e.g. mixing and residence time);
- Nutrient regulation and stoichiometry of algal biomass (i.e. phytoplankton and macroalgae) production including nutrient related selection of dominant species, functional groups, and algal community structure;
- Development of new phytoplankton assessment tools that account for shifts in species composition and frequency of blooms in the status assessment;
- Relationship between nutrient enrichment and shifts in structure and functioning of the planktonic food web;
- Development of monitoring tools that account for rapid changes in algal communities, allowing detection of bloom peaks (e.g. continuous measurements, ships-of-opportunity, remote sensing tools, algorithm development, etc.);
- Effect of top-down control (e.g. shellfish filtration, zooplankton grazing) and other food web interactions (viral infections, parasitism, including the role of mixotrophy (ability to use organic sources of N and P) etc) in regulation of algal biomass and transmitted/amplified effects.

6.3. Harmful algal blooms

- Identification and understanding of the link between HABs and land based nutrient inputs;
- Identification of the role of mechanisms such as upwelling relaxation events, cyst formation etc in HAB formation, and the extent to which these events are manageable

6.4. Value, resilience and recovery of marine ecosystems

- Marine submerged vegetation is valuable for maintenance of biodiversity as it forms habitat for many organisms (invertebrates, fish juveniles, etc.). Research is needed on evaluation of eutrophication impacts including the optimal extent and status of these communities for supporting viable and diverse communities; valuation of goods and services provided by such communities and development of tools for marine spatial planning and management of marine protected areas with respect to eutrophication are also an important area for research;
- Identification of factors that govern the occurrence and extension of the hypoxic/anoxic events as well as the impacts of such events on resilience and recovery of benthic communities. There is a need to distinguish between the natural range and increases in spatial extent of anoxic sediments and bottom waters due to anthropogenic organic loading;
- Determination of the resilience of marine ecosystems for identification of critical nutrient loading thresholds beyond which the whole system shifts to an alternative steady state. This includes research exploring potential recovery pathways from eutrophic to non-eutrophic states. This is not well established because system functioning and components may have changed and the recovery pathway and restoration outcome may not be identical to rate of deterioration or the original status before impairment.

6.5. Selection of criteria and indicators for eutrophication assessment by the MSFD

The efforts of the working group on the MSFD qualitative descriptor of ’human-induced eutrophication’ resulted in the selection by the European Commission (2010) of three different aspects (nutrient levels; direct and indirect effects of nutrient enrichment) and eight indicators, which can potentially be used in the environmental status assessment within the MSFD:

(a) for nutrient levels: nutrient concentration in the water column; nutrient ratios (silica, nitrogen and phosphorus);

(b) for direct effects of nutrient enrichment: chlorophyll concentration in the water column; water transparency related to increase in suspended algae; abundance of opportunistic macroalgae; and species shift in floristic composition, such as diatom to flagellate ratio, benthic to pelagic shifts, as well as bloom events of nuisance/toxic algal blooms caused by human activities;
(c) for indirect effects of nutrient enrichment: abundance of perennial seaweeds and seagrasses adversely impacted by decrease in water transparency; dissolved oxygen changes due to increased organic matter decomposition and size of the area concerned.

The assessment must consider relevant temporal scales and the relationship to nutrient loads from rivers in the catchment area. The EC decision on criteria and methodological standards on good environmental status of marine waters (European Commission, 2010) encourages the use of previous information and knowledge gathered and approaches developed in the framework of regional sea conventions, such as those described here, as a starting point.

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