How well do ecosystem indicators communicate the effects of anthropogenic eutrophication?

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ABSTRACT

Anthropogenic eutrophication affects the Mediterranean, Black, North and Baltic Seas to various extents. Responses to nutrient loading and methods of monitoring relevant indicators vary regionally, hindering interpretation of ecosystem state changes and preventing a straightforward pan-European assessment of eutrophication symptoms. Here we summarize responses to nutrient enrichment in Europe's seas, comparing existing time-series of selected pelagic (phytoplankton biomass and community composition, turbidity, N:P ratio) and benthic (macro flora and faunal communities, bottom oxygen condition) indicators based on their effectiveness in assessing eutrophication effects. Our results suggest that the Black Sea and Northern Adriatic appear to be recovering from eutrophication due to economic reorganization in the Black Sea catchment and nutrient abatement measures in the case of the Northern Adriatic. The Baltic is most strongly impacted by eutrophication due to its limited exchange and the prevalence of nutrient recycling. Eutrophication in the North Sea is primarily a coastal problem, but may be exacerbated by climatic changes. Indicator interpretation is strongly dependent on sea-specific knowledge of ecosystem characteristics, and no single indicator can be employed to adequately compare eutrophication state between European seas. Communicating eutrophication-related information to policy-makers could be facilitated through the use of consistent indicator selection and monitoring methodologies across European seas. This work is discussed in the context of the European Commission's recently published Marine Strategy Directive.

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1. Introduction

Eutrophication is widely recognised as a key problem affecting Europe's seas in the technical reports and policy statements of the OSPAR (OSPAR Commission, 2000), Helsinki (HELCOM, 1991), and Black Sea Commissions (Black Sea Commission, 1996), the Mediterranean Action Plan (MAP – UNEP, 1996), and the European Environment Agency (EEA – Ærtebjerg et al., 2001). Additionally, the minimisation of eutrophication effects is specifically mentioned

as a requirement of good environmental status in the European Union's Marine Strategy Directive (European Commission, 2008). A causal link between anthropogenic sources of nutrients and the emergence of eutrophication symptoms is generally accepted (Ærtebjerg et al., 2001; Smith, 2006). However, cause–effect relationships are not straightforward as coastal ecosystems respond to nutrient loading in various ways. System-specific attributes may act as a filter to modulate responses to enrichment and a complex suite of direct and indirect responses may interact (Cloern, 2001); in the Black Sea, for example, the combined effects of nutrient loading and overfishing resulted in a trophic cascade which altered the ecosystem's structure and dynamics (Daskalov, 2002). Other sources of environmental degradation, such as toxic substances, overfishing, and invasive species, as well as climate and natural variability, may confound this causality (Caddy, 2000; MacKenzie et al., 2002; Nixon and Buckley, 2002; MacKenzie and Koster, 2004; Oguz, 2005;

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McQuatters-Gollop et al., 2007). Our growing understanding of anthropogenic impact on coastal systems (e.g. Cloern, 2001; Elmgren, 2001) includes recognition of non-linear responses and even regime shifts by entire ecosystems (e.g. Beaugrand, 2004).

From 2012 as a condition of the Marine Strategy Directive, EU Member States must monitor relevant eutrophication indicators in their waters; these indicators are required to be comparable between regions (European Commission, 2008). Monitoring programmes, some underway for decades (Cociasu and Popa, 2004; Wiltshire and Manly, 2004; Richardson et al., 2006), currently record a range of relevant variables for assessment and periodic reporting on the state of Europe's marine environment. This paper examines a selection of commonly reported indicators of ecosystem state, focusing on those used to monitor marine eutrophication. The indicators examined here can be found in Annex III of the Marine Strategy Directive which includes a list of physical, chemical and biological indicators suggested for use in monitoring progress towards good environmental status of marine waters (European Commission, 2008). The aim of our research is therefore to answer the following question: Do the eutrophication-relevant ecosystem indicators suggested in the Marine Strategy Directive provide consistent, scientifically founded information to European policy-makers so that they can understand and compare eutrophication status in Europe's regional seas?

Nutrient enrichment generates two primary, interrelated effects in aquatic ecosystems: firstly, stimulation of phytoplankton growth, and in some cases a change in phytoplankton species composition, favouring opportunistic and even harmful species, in the pelagic zone; and secondly, shading and deposition of organic matter in the benthic zone. Our selection of ecosystem indicators distinguishes between pelagic and benthic ecosystems to capture these primary effects. From the large range of indicators and variables reported in the literature (e.g. Gazeau et al., 2004, which also offers a comparison of Europe's regional seas), we confine ourselves to those for which long data series exist for regions of Europe's seas that suffer

from eutrophication: the coastal North Sea, the Baltic Proper, the Northern Adriatic Sea, and the northwest shelf of the Black Sea (Fig. 1). Following a short description of our method, this paper summarizes responses to nutrient enrichment in the regional study areas. We then compare the indicators presented based on their effectiveness in assessing eutrophication effects, and draw implications for policy.

2. Methods

The work presented here was part of the EU FP6-funded European Lifestyles and Marine Ecosystems (ELME) project. One of the objectives of ELME was to gather as much information as possible on well established ecosystem trends that could be used for the future management of Europe's seas. ELME used an 'indicator' approach to exploring change in Europe's seas; although predictive models are highly desirable for management purposes, they must be fed with data, which have their own intrinsic value. Eutrophication was a priority issue of the project and a key product was the aggregation and analysis of existing relevant long-term datasets in European marine and coastal regions where eutrophication is a historical concern. Many of the datasets gathered during ELME coincide with the indicators listed in the Marine Strategy Directive.

2.1. Areas of study

The Baltic Proper forms the central and largest basin of the Baltic Sea. The Baltic Proper is brackish, with a distinct north-to-south salinity gradient, and is the most limited in exchange of the four study areas. Nutrients entering the Baltic have long residence times: between 4.4 and 22.5 years for P in the Baltic Proper (Savchuk, 2005). The Baltic Sea's catchment can be divided into a northern boreal part draining into the Gulf of Bothnia and a south-eastern part draining into the Baltic Proper (Savchuk, 2005). The latter is predominantly agricultural. Rivers, especially
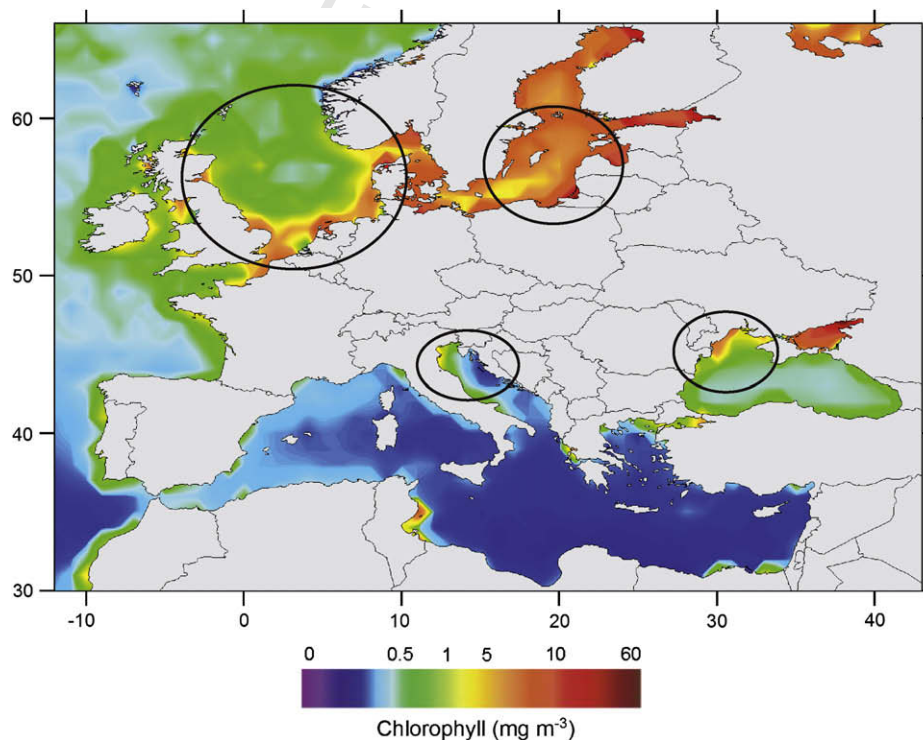

Fig. 1. SeaWiFS (Sea-viewing Wide Field-of-View sensor) remote sensing images showing chlorophyll concentrations in four European regional seas: Baltic, Mediterranean, Black and North – annual composite for 2007. SeaWiFS overestimates chlorophyll in waters with high-suspended substances, such as some coastal areas and the Baltic Sea.

the Oder and Vistula draining Poland with its 40 million inhabitants, and the Daugava and Nemunas, are the largest contributors of anthropogenic nutrients (Stalnacke et al., 1999). Major cities such as St. Petersburg, Helsinki and Stockholm are coastal point sources of nutrients (HELCOM, 2006). In the Baltic, eutrophication is an open water as well as coastal phenomenon (Elmgren, 2001). In open waters, benthic and pelagic environments are coupled via a positive feedback linking cyanobacterial blooms, bottom hypoxia and sedimentary reflux of P (Conley et al., 2002; Vahtera et al., 2007). This feedback partly inhibits recovery from eutrophication (Vahtera et al., 2007). In coastal waters, the effects of eutrophication are evident in the depth limitation of macrophytes, particularly *Zostera marina* (Duarte, 1991; Nielsen et al., 2002; Krause-Jensen et al., 2005), and increased macrozoobenthos biomass where flushing prevents bottom hypoxia (Karlson et al., 2002; Perus and Bonsdorff, 2003).

Of the four seas, the coastal North Sea has the highest flushing rate (water residence time is only 0.9 months, calculated from Lenhart et al. (1995)) and inflow from the northeastern Atlantic Ocean is the predominant source of nutrients. Rivers such as the Rhine and Elbe, which drain much of northwestern Europe, are the primary source of anthropogenic nutrients. Much has been done to reduce riverine nutrient loads (Ærtebjerg et al., 2001). The main sources at present are wastewater treatment plants that do not remove nutrients from their effluents, and agriculture with its use of fertilizers and production of manure (Ærtebjerg et al., 2001). Riverine nutrient loads fuel phytoplankton growth in coastal waters and in the shallower, relatively poorly flushed southern North Sea. Hypoxia has occurred in some near-shore areas, notably the German Bight (Brockman et al., 1988; Kronvang et al., 1993); hypoxia is often associated with monospecific blooms of organisms such as *Phaeocystis* that cause major ecosystem effects (Lancelot et al., 2006). Additionally, coastal bays in the North Sea are becoming increasingly prone to filamentous macroalgal blooms (Krause-Jensen et al., 2007). However, eutrophication effects on benthic environments in many regions of the coastal North Sea are probably overshadowed by damage from trawling (Rijnsdorp et al., 1998; Callaway et al., 2007).

Compared with the North, Black and Baltic Seas, the Mediterranean is oligotrophic. However, the Northern Adriatic is one of the most productive areas in the Mediterranean. It is a shallow (average depth 35 m) enclosed basin with moderate flushing with a residence time of 3.3 months, calculated from Zavatarelli and Pinardi (2003). The Northern Adriatic receives considerable fresh water discharge (about 80 km^3 yr^{-1}), mainly from Italian rivers (e.g. Po, Adige, Isonzo, Tagliamento) that drain intensely developed catchments where about 50% of Italy's national production of agriculture and livestock, and 40% of its population, are concentrated (Pirrone et al., 2005). These rivers discharge high nutrient loads making the Northern Adriatic the most vulnerable area of the Mediterranean regarding eutrophication (UNEP, 1996). In the 1980s, during the period of peak discharge, the basin suffered severe eutrophication with recurrent red tides (Marasovic et al., 1991,2005) and hypoxia and anoxia crises (Vollenweider et al., 1992; Degobbis et al., 2000), resulting in fish kills (Marasovic et al., 1991).

The Black Sea has very limited exchange with a residence time of ~2000 years for the whole sea (Ozsoy and Unluata, 1997) and 12 months for the northwest shelf (calculated from Gregoire and Friedrich (2004)). However, it has a large, permanently anoxic (below approximately 100 m), deepwater sink that serves to remove nutrients from further biological activity. Eutrophication symptoms are most pronounced in the relatively shallow northwest shelf that comprises approximately 12% of the surface area of the Black Sea. This shelf receives the discharge of Europe's second and third largest rivers (the Danube and Dnieper), together draining large parts of 12 countries. Increased use of fertilizers and phosphate detergents, fossil fuel combustion and sewage reticulation without nutrient removal led to loading of these rivers and subsequent problems with eutrophication on the shelf (Mee, 1992). Summer hypoxia as a result of eutrophication extended over most of the shelf in the 1980s (Zaitsev and Mamaev, 1997). These hypoxic events had become so severe in the late 1980s that huge quantities of benthic organisms died and were washed up to decay on the beaches of Romania and Ukraine (Zaitsev and Mamaev, 1997). Eutrophication-related effects in shelf waters have lessened, largely due to post-Soviet economic decline that has reduced nutrient inputs (Mee, 2006).

2.2. Data availability and selection of ecosystem indicators

Both ELME and the Marine Strategy Directive distinguish between indicators of ecosystem 'state' (referred to as 'characteristics' in the Marine Strategy Directive) and indicators of anthropogenic 'pressures' leading to ecosystem change (Langmead et al., 2007; European Commission, 2008). This paper focuses on eutrophication-related state indicators rather than those representing pressures. Although pressure indicators such as nutrient loads, can potentially provide an early warning of eutrophication, there has been limited investment in the science behind their use as indicators, and quantitative links still need to be drawn between pressures and state changes. While some advances in predictive modelling have been made (Vollenweider et al., 1998; examples in **Q1** Smith, 2003; Lancelot et al., 2006; Smith, 2006), these models are often complex and the ecosystem responses to changes in nutrients are system specific and difficult to generalize. Research into the response of ecosystem state indicators to increased nutrients, however, is further developed and representative time-series are more widely available. While pelagic state indicators (such as phytoplankton biomass or community composition) may provide an early warning of eutrophication-induced change, indicators of benthic state (such as hypoxia), although frequently used to assess eutrophication, often provide information too late for response (Mee, 2005). Thus the final selection of indicators per regional sea, shown in Table 1, is a compromise among data availability (long time-series), frequency of use in the literature, and sea-specific features. Data were primarily obtained from published literature, but a substantial effort was put into collecting and using datasets which are unpublished or only published in the grey literature. In many cases the data used here are also freely available on the internet as a result of monitoring programmes. In general, data for the North and Baltic Seas are much more readily available than those for the Black and Adriatic Seas where few long-term monitoring programmes exist. Unfortunately, monitoring programmes in the four study areas do not always collect similar data in the same way.

The effects of nutrient enrichment can manifest in both pelagic and benthic ecosystems. For the pelagic ecosystems, we identified four general indicators: phytoplankton biomass (usually measured as chlorophyll concentration) to capture enhanced phytoplankton growth; Secchi depth, a measure of turbidity which addresses the effect of enhanced phytoplankton growth on light penetration; phytoplankton community composition; and N:P ratio to assess possible effects of imbalanced nutrient enrichment on the phytoplankton. Nutrient ratios in coastal systems can differ markedly from the Redfield ratio (Cloern, 2001), the average molecular ratio at which phytoplankton require these two elements (Redfield et al., 1963) and this may have an effect on pelagic food webs (Philippart et al., 2000).

Because of the lack of coordination between observation systems amongst Europe's seas, different measures of

Table 1

Ecosystem state indicators for eutrophication arising from the ELME project. Indicators were selected from existing datasets and were chosen based on data availability, relevance to study areas and frequency of use in the literature.

Study area	Pelagic ecosystem		Benthic ecosystem	
	Indicator (units)	Reference identifying data source	Indicator (units)	Reference identifying data source
Coastal North Sea	Secchi depth (m)	McQuatters-Gollop et al. (2007)	None	No benthic indicators for eutrophication were selected for the North Sea as the effects of eutrophication are probably overshadowed by those of trawling (Rijnsdorp et al., 1998; Callaway et al., 2007)
	Phytoplankton biomass (annual chlorophyll a, mg m^{-3})	McQuatters-Gollop et al. (2007)		
	Proportion diatoms (ratio of abundance of diatoms relative to dinoflagellates)	Data from the Continuous Plankton Recorder survey (SAHFOS, 2004)		
	N:P (molar ratio)	McQuatters-Gollop et al. (2007)		
	Annual mean TN, TP (μM l^{-1})	McQuatters-Gollop et al. (2007)		
Baltic Proper	Secchi depth (m)	Aarup (2002)	Max. depth of *Zostera marina* (m)	Adapted from Langmead et al. (2007)
	N:P (molar ratio)	(MARE – Marine Research on Eutrophication)	Area of hypoxia (summer, km^{-2})	Conley et al. (2002)
	Phytoplankton biomass (summer chlorophyll a, μg l^{-1})	ICES (2008)		
	Annual mean TN, TP pool (metric tons)	(MARE – Marine Research on Eutrophication)		
Black Sea NW shelf	Secchi depth (m)	NIMRD (2008)	Area of hypoxia (summer, km^{-2})	Adapted from Mee (2006)
	Phytoplankton biomass (summer wet weight biomass, mg m^{-3})		*Mytilus galloprovincialis* biomass (g m^{-2})	Langmead et al. (2009)
	Proportion diatoms (ratio of abundance of diatoms relative to dinoflagellates)			
	N:P (molar ratio)		Area of *Phyllophora* field (km^{-2})	Langmead et al. (2009)
	Annual mean DIN, DIP (μM l^{-1})			
Northern Adriatic	Secchi depth (m)	ARPAER (2006)	Bottom dissolved oxygen (summer, ml l^{-1})	ARPAER (2006)
	Phytoplankton biomass (summer chlorophyll a, μg l^{-1})	Adapted from ARPAER (2006) and EEA (2006)		
	Proportion diatoms (ratio of abundance of diatoms relative to dinoflagellates)	Adapted from Mazziotti et al. (2005)		
	N:P (molar ratio)	Adapted from Mazziotti et al. (2005)		
	Annual mean DIN, TP (μM l^{-1})	Adapted from Mazziotti et al. (2005)		

phytoplankton biomass were used in each study area. While we present chlorophyll data (μg l^{-1}) for the Baltic Proper in Fig. 2a, indicators of phytoplankton biomass are of limited applicability in the Baltic Sea. High levels of dissolved organic carbon in Baltic waters interfere with determination of chlorophyll concentrations by remote sensing (Darecki and Stramski, 2004) whereas in situ measurement of biomass is complicated by the non-uniform distribution in the water column of cyanobacteria, the dominant summer phytoplankton group in the low saline Baltic Sea (Kutser, 2004). Therefore, the chlorophyll concentrations indicated in Fig. 1 and based on remote sensing are very likely to be an overestimate. Data on the extent and duration of cyanobacterial blooms were available for 1997–2007 (Hanson, 2007) but, at only 10 years, this time-series was too short to assess trends. For the Baltic Proper, Secchi depth is an often used proxy for phytoplankton biomass (Erlandsson and Stigebrandt, 2006; HELCOM, 2006). For the North Sea, annual mean chlorophyll a (mg m^{-3}) was derived from the relationship between in situ phytoplankton biomass and remotely sensed chlorophyll (see McQuatters-Gollop et al., 2007). In situ measured summer chlorophyll a (μg l^{-1}) is used in the Adriatic, and summer phytoplankton biomass (mg m^{-3}, wet weight) in the Black Sea.

Data on phytoplankton community composition, expressed as the ratio of abundance of diatoms relative to abundance of dino-flagellates, were available for the North, Black and Northern Adriatic Seas. Phytoplankton community composition data for the North Sea are from the Continuous Plankton Recorder (CPR – SAHFOS, 2004) which records only large plankton and armoured flagellates and underestimates abundance of other plankton groups (such as nanoplankton or naked flagellates). However, the proportion of cells captured by the silk reflects the major changes in abundance, distribution, and community composition of the phytoplankton, and, most importantly, is consistent and comparable over time (Batten et al., 2003; Richardson et al., 2006). Data are available on phytoplankton composition in the Baltic, but because a link between these trends and eutrophication has been deemed uncertain, they are not assessed here (Jaanus et al., 2007). Time-series measuring Secchi depth, nutrient concentrations (as total nitrogen (TN), total phosphorus (TP), dissolved inorganic nitrogen (DIN), and/or dissolved inorganic phosphorus (DIP)) and molar N:P ratio were available for all four regional seas. As mentioned above, there is regional variability in ecosystem response to nutrient loading, and limited investment into the use of nutrient concentrations themselves as indicators of eutrophication. Consequently, this work focuses on ecosystem changes in response to changing nutrient availability. We offer no in depth exploration of change in nutrient concentrations and the reasons behind their dynamics, as this information is available elsewhere (see Ærtebjerg et al., 2001; EEA, 2005; Artioli et al., 2008; Vermaat et al., 2008). However time-series of nutrient concentrations are presented together with the other state indicators in order to increase the usefulness of this paper.

For benthic ecosystems, we assessed benthic floral and faunal communities as well as bottom oxygen conditions. Depth limitation of macrophyte communities has been found to demonstrate the effects of light attenuation in the pelagic zone (Duarte, 1991; Nielsen et al., 2002; Krause-Jensen et al., 2005) while their spatial extent can be linked to benthic oxygen conditions (Zaitsev and Mamaev, 1997). Macrozoobenthos can benefit from an increased 'rain' of pelagic-derived organic matter, but suffer under conditions of hypoxia (Pearson and Rosenberg, 1978). No benthic indicators were selected for the North Sea due to the widespread impact of
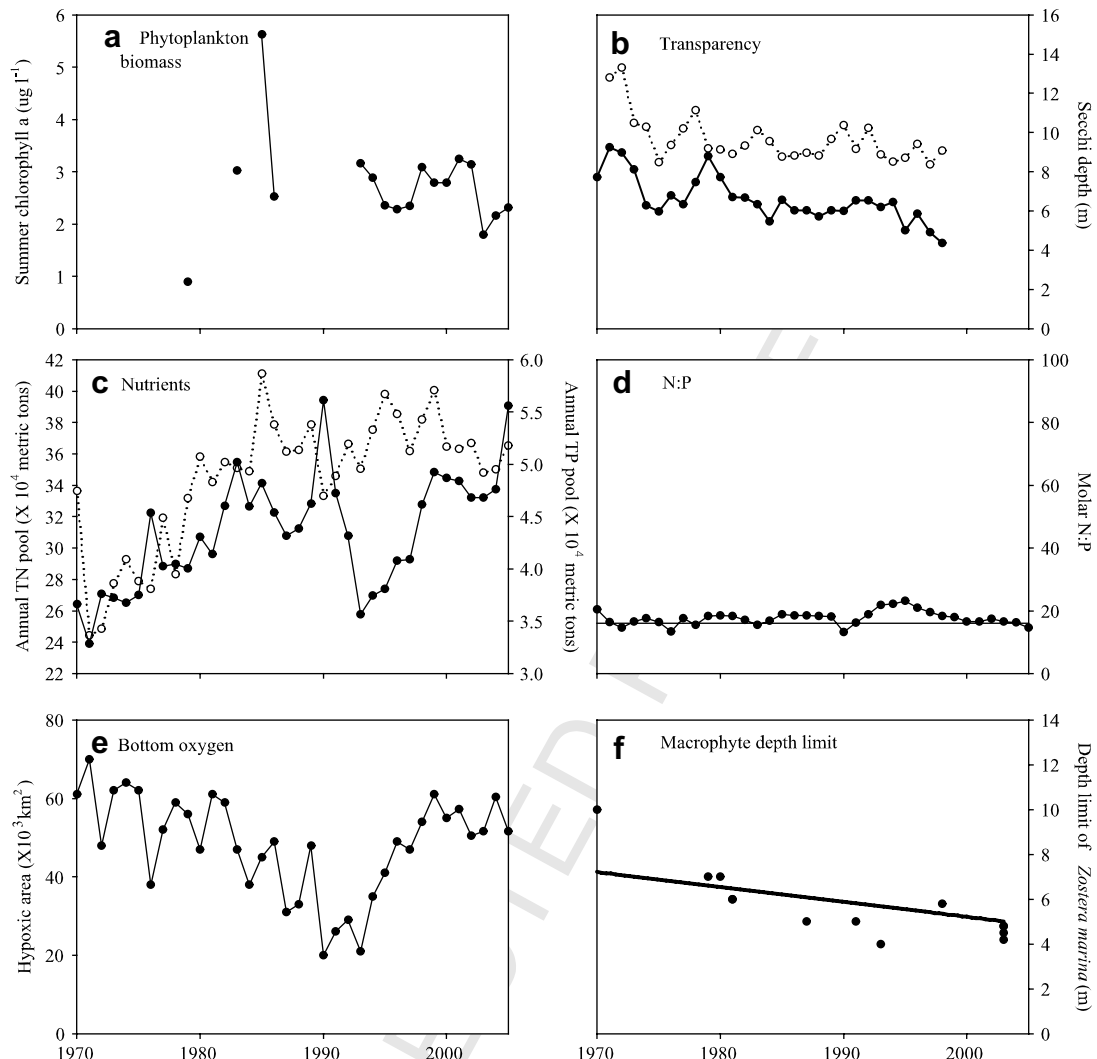
Fig. 2. Trends in pelagic and benthic eutrophication indicators in the Baltic Proper: (a) phytoplankton biomass as indicated by mean summer chlorophyll a concentration; (b) water transparency (and, indirectly, cyanobacteria biomass) as indicated by summer (●) and winter (○) Secchi depth; (c) mean annual pool of TN (○) and TP (●); (d) molar ratio of N:P, the solid line denotes the molar Redfield ratio of 16:1; (e) bottom oxygen as indicated by area of benthic hypoxia; and (f) macrophyte depth limit as indicated by maximum depth of *Zostera marina* ($r = 0.84$, $p < 0.001$).

trawling (Callaway et al., 2007). Indicators regarding benthic macrophytes vary between seas. A meta-data analysis of information derived from the literature assessed changes in depth limitation of eelgrass (*Z. marina*) in the Baltic Proper. Spatial extent of the Black Sea's *Phyllophora* field was selected to illustrate the impacts of hypoxia on this biologically diverse benthic community. Comparable macrophyte time-series were unavailable for the Northern Adriatic.

A time-series reflecting eutrophication effects on macrozoobenthic biomass was only available for the Black Sea. A metadata analysis of *Mytilus galloprovincialis* biomass data was assessed for the northwest shelf. Zoobenthos data for the Baltic Proper are not evaluated here as much of the Baltic Proper has been primarily hypoxic or anoxic since the 1960s with no or reduced zoobenthos (Karlson et al., 2002). For the Adriatic, too few data were available to estimate quantitative trends in zoobenthos biomass; furthermore Scardi et al. (1997) suggest suspended sediment, rather than eutrophication, as the primary driver of change in zoobenthos composition.

Different ways of reporting bottom oxygen measurements were also encountered – hypoxic area (area with oxygen $< 2\,\mathrm{ml\,l^{-1}}$) is

used as an indicator of bottom oxygen in the Baltic and Black Seas while dissolved oxygen concentration ($\mathrm{ml\,l^{-1}}$) is the indicator used in the Northern Adriatic (Table 1).

3. Results

3.1. Pelagic responses

Partially due to its low salinity, the strongest response to eutrophication in the open Baltic Proper occurs in the N-fixing cyanobacteria, blooms of which have become common since the 1960s (Finni et al., 2001). This trend is approximated here by a declining Secchi depth (Fig. 2b, see also Fleming-Lehtinen et al., 2007). Nutrients in the Baltic have undergone a series of trends. Both TN and TP increased during 1970s and 1980s, before a large drop in TP occurred during the early 1990s, after which TP steadily increased (Fig. 2c). The N:P ratio (Fig. 2d) was near the 'ideal' Redfield ratio of 16:1, but has shown a distinct decrease since 1994, indicating an increasing availability of P relative to N. This supports the notion that nutrient conditions are increasingly favouring cyanobacteria, possibly via the positive feedback of P reflux linking

benthic and pelagic environments (Smith, 1983; Kahru et al., 2000; Vahtera et al., 2007). Due to this feedback, Baltic open water pelagic conditions cannot be discussed separately from trends in bottom hypoxia. Bottom hypoxia in the Baltic shows a declining trend from the 1970s to the early 1990s (Fig. 2e). This corresponds to a so-called stagnation period with no major saline incursions from the North Sea, but also a progressive downward erosion of the halocline leading to oxygenation of bottom sediments (Gerlach, 1994; Conley et al., 2002).

In the coastal North Sea, phytoplankton biomass has shown an increasing trend since the 1980s (Fig. 3a). Secchi depth has been increasing since the mid-1970s (Fig. 3b), a change that has been linked to an increase in Atlantic inflow into the North Sea (McQuatters-Gollop et al., 2007). This apparent paradox – higher phytoplankton biomass yet greater water clarity – is exacerbated by declining nutrient concentrations (Fig. 3c). Coastal chlorophyll and water clarity have been positively correlated, while chlorophyll and TN and TP concentrations are negatively correlated (see also McQuatters-Gollop et al., 2007). Additionally, a positive correlation was observed between annual mean chlorophyll a and annual mean dissolved silica (Si) concentration (data not shown). The increase in phytoplankton biomass since the mid-1980s,

and particularly of the diatom fraction since the early 1990s (Fig. 3a and e), therefore may be related to greater availability of Si (Officer and Ryther, 1980; Humborg et al., 2000; Wirtz and Wiltshire, 2005), as well as decreased turbidity. The annual molar ratio of N:P has consistently been greater than the Redfield ratio since at least 1980 (Fig. 3d). The increase in N:P is partially an artefact of the similar rate of decline of TN and TP concentrations (Fig. 3e). A peak ratio of nearly 60 in the early 1980s has since been followed by ratios fluctuating around 30. The highest value of this ratio coincided with a low proportion of diatoms although there is no significant statistical correlation between N:P ratio and diatom fraction, however, other workers have also observed a link between changes in nutrient ratios and phytoplankton community composition in the coastal North Sea (Philippart et al., 2000; Lancelot et al., 2006, among others).

Chlorophyll data for the Northern Adriatic show two peaks – around 1988 and 1998 – with a strong decline since 2000 (Fig. 4a). Secchi depth data (Fig. 4b) suggest a decrease during the 1990s followed by an increase around 1998. Trends in phyto-plankton could explain trends in turbidity, but the data series is too short for any certainty here. Further, the Northern Adriatic is consistently much more turbid than the other study areas. Given
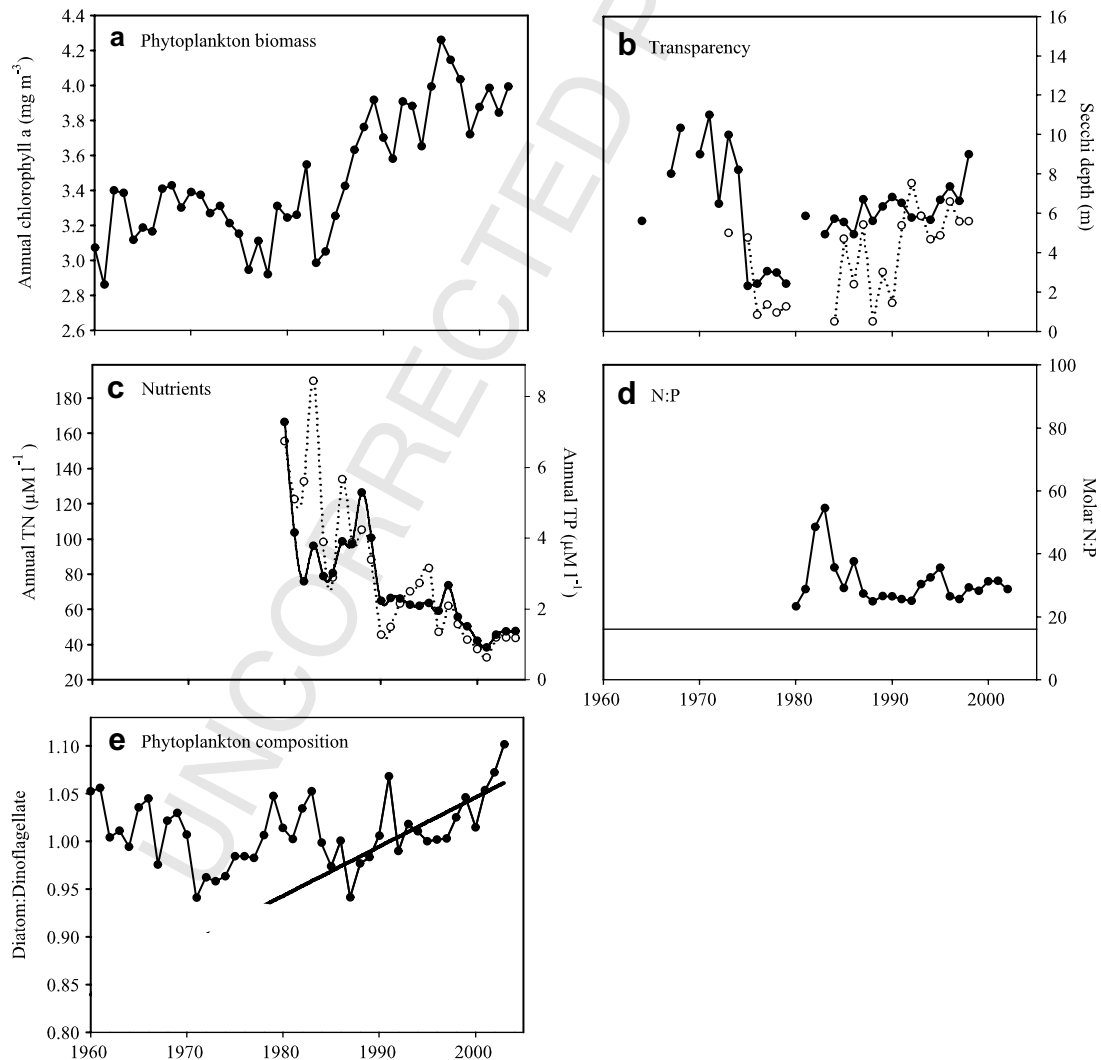
Fig. 3. Trends in pelagic and benthic eutrophication indicators in the coastal North Sea: (a) phytoplankton biomass as indicated by mean annual chlorophyll a concentration; (b) water transparency as indicated by summer (●) and winter (○) Secchi depth; (c) mean annual concentrations of TN (○) and TP (●); (d) molar ratio of N:P, the solid line denotes the molar Redfield ratio of 16:1; (e) phytoplankton community composition as represented by the ratio of abundance of diatoms to dinoflagellates ($r = 0.75$, $p < 0.001$, from 1980 onwards).
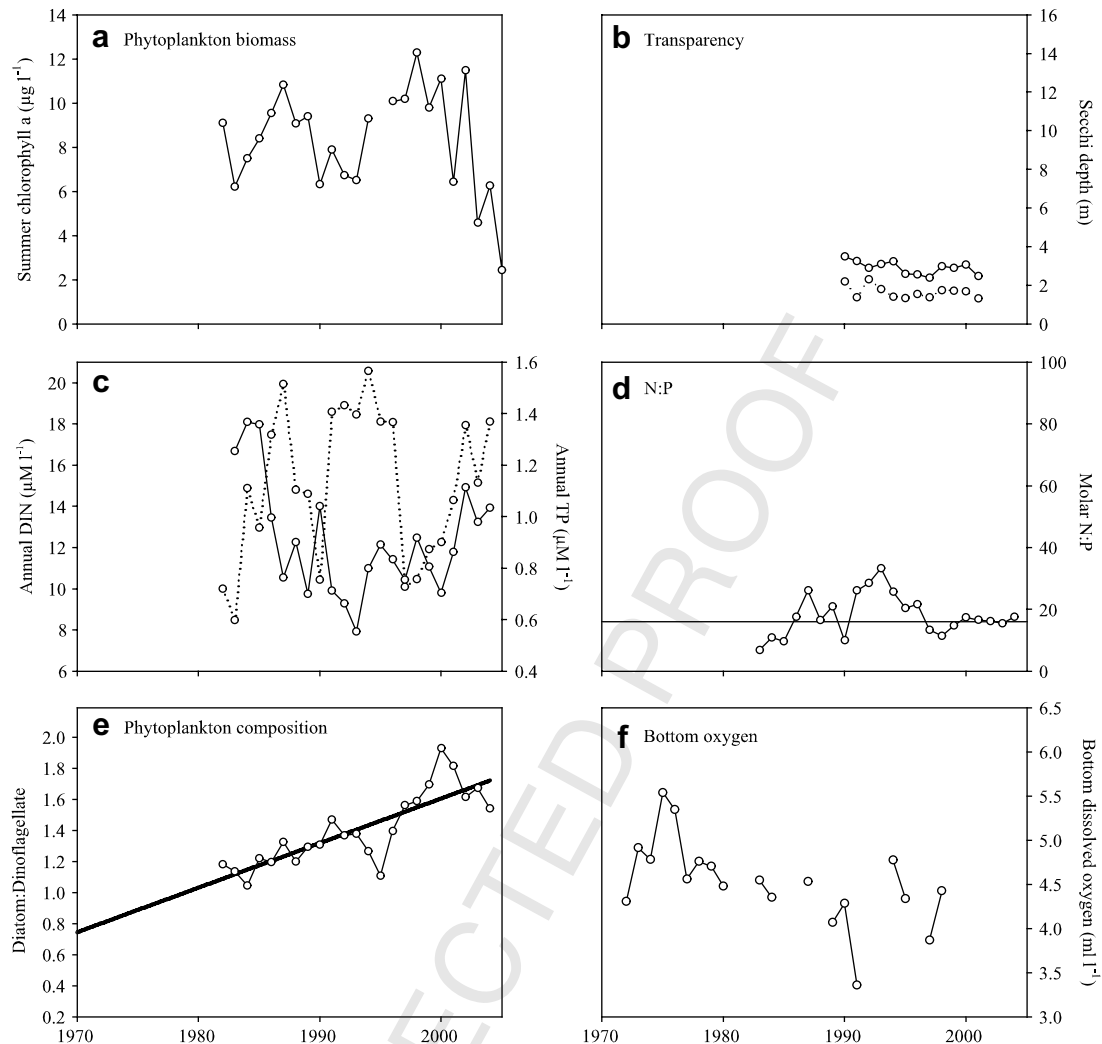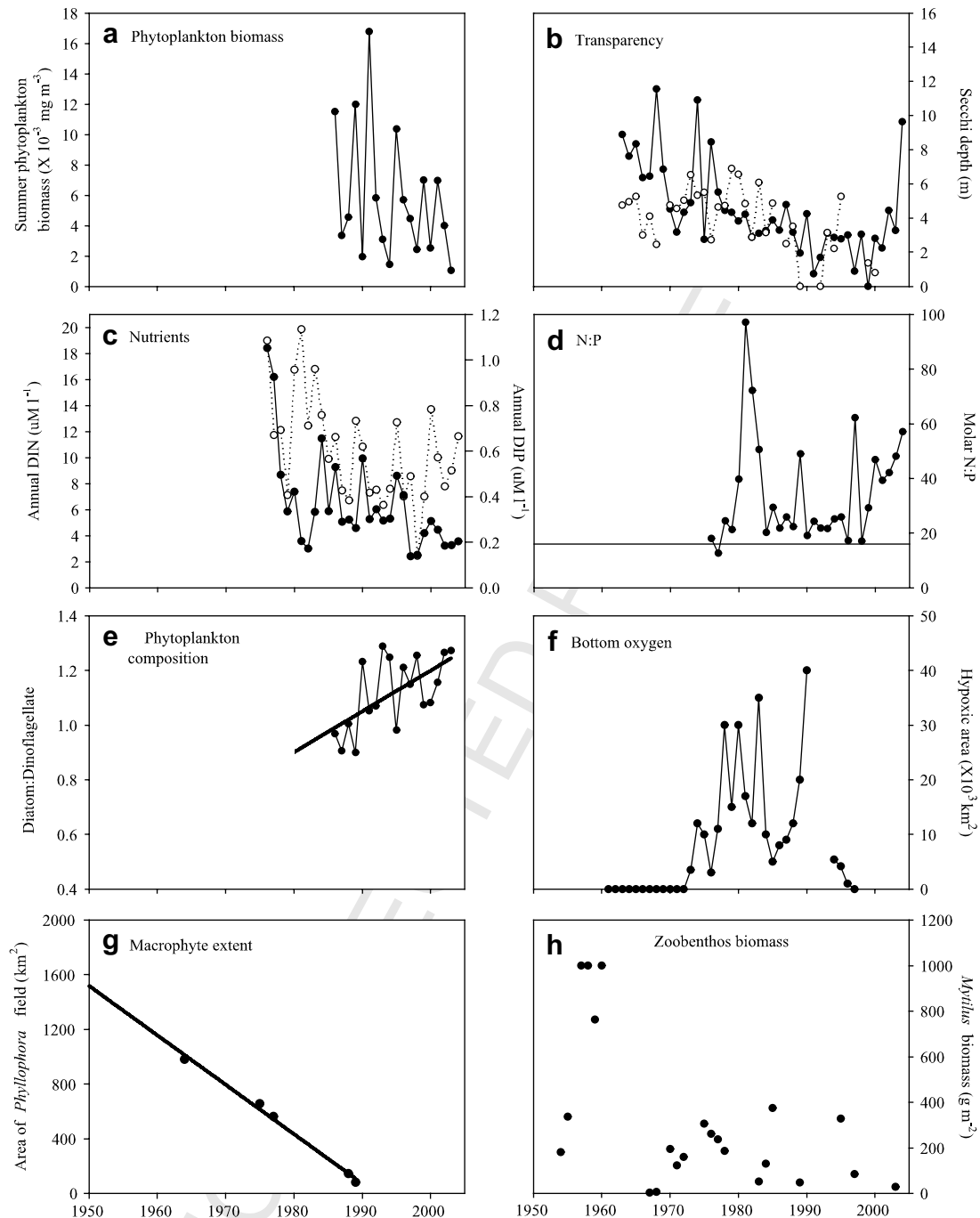
Fig. 4. Trends in pelagic and benthic eutrophication indicators in the Northern Adriatic: (a) phytoplankton biomass as indicated by mean summer chlorophyll a concentration; (b) water transparency as indicated by summer (●) and winter (○) Secchi depth; (c) mean annual concentrations of DIN (○) and TP (●); (d) molar ratio of N:P, the solid line denotes the molar Redfield ratio of 16:1; (e) phytoplankton community composition as represented by the ratio of abundance of diatoms to dinoflagellates ($r = 0.82$, $p < 0.001$); (f) bottom oxygen as indicated by bottom dissolved oxygen concentration.

that it is also the shallowest, suspended sediments can be expected to interfere with a correlation between phytoplankton and turbidity, as is the case in the coastal North Sea (McQuatters-Gollop et al., 2007). Nutrient concentrations showed no clear trend but were variable throughout the time-series, also due to the high influence of water–sediment interaction in those near-shore stations (Fig. 4c). After high, but variable, N:P ratios between 1985 and 1995, N:P has since returned to values close to the Redfield ratio (Fig. 4d). With the exception of the early 1990s, the ratio of diatoms to dinoflagellates has been increasing since the early 1980s (Fig. 4e). The steep decline in this proportion after 1998, coinciding with a general decrease in phytoplankton biomass, is difficult to explain. The decrease in phytoplankton biomass, shift in phytoplankton community composition towards diatoms and decreased N:P ratio may suggest a declining impact of eutrophication in the Northern Adriatic pelagic ecosystem despite the unclear trends in turbidity.

On the northwest Black Sea shelf, the 1970s and 1980s were characterised by high phytoplankton biomass and an increased number of mass blooming species, most of which were flagellates, especially dinoflagellates, euglenophytes and prymnesiophytes (Bodeanu et al., 2004). The steady decline in water transparency

from the 1960s (Fig. 5b) inversely approximates the trend in phytoplankton biomass (Fig. 5a). The change from diatom domi-nated phytoplankton to these non-siliceous species coincided with decreasing trends in Si loads of the Danube, whereas N and P loads increased during the 1970s and 1980s (Humborg et al., 1997). Since the mid-1980s, nitrogen and phosphorus concentrations have declined in Black Sea waters (Fig. 5c) Molar N:P ratio was high during the early 1980s suggesting relatively greater N loading (Fig. 5d). During the 1990s it was slightly above the Redfield ratio, but during the last decade N:P ratio has shown an increasing trend. Should there be a relationship between diatom fraction and N:P ratio, the data series is too short to suggest what it might be, however diatom and dinoflagellate abundances have decreased in the Black Sea while abundance of 'other' phytoplankton groups, such as cyanophytes and haptophytes, has increased (Bodeanu et al., 2004). Since 2000, coincidence in the decline in phyto-plankton biomass (Fig. 5a), increase in Secchi depth (Fig. 5b) and increase in diatom fraction (Fig. 5e) supports claims that the shelf has entered a period of recovery (Bodeanu et al., 2004; Mee et al., 2005). Bodeanu et al. (2004) further argues that the phytoplankton community has returned to a composition similar to that found before 1970.

Fig. 5. Trends in pelagic and benthic eutrophication indicators in the NW Black Sea shelf: (a) phytoplankton biomass as indicated by mean summer phytoplankton biomass; (b) water transparency as indicated by summer (●) and winter (○) Secchi depth; (c) mean annual concentrations of DIN (○) and DIP (●); (d) molar ratio of N:P, the solid line denotes the molar Redfield ratio of 16:1; (e) phytoplankton community composition as represented by the ratio of abundance of diatoms to dinoflagellates ($r = 0.61$, $p < 0.01$); (f) bottom oxygen as indicated by hypoxic area; (g) macrophyte extent as represented by spatial area of *Phyllophora* field ($r = 0.99$; $p < 0.001$); and (h) zoobenthos biomass as indicated by the biomass of *Mytilus galloprovincialis*.

3.2. Benthic responses

In the Baltic Proper the depth limits of *Z. marina* have clearly decreased since the beginning of the 20th century from 6–10 m (8.7 ± 0.7 m on average) during the 1950s to 1980s, to 4–6 m (5.1 ± 0.4 m on average) in contemporary times (Fig. 2f, see also Nielsen et al., 2002). The data do not indicate a levelling off of this trend. The change in depth range of *Zostera* occurrence appears to be related to decreasing Secchi depth in the Baltic Proper. This suggests that shading by pelagic algae is a key factor in depth

limitation of this species (Fig. 2, also Nielsen et al., 2002). Trends in hypoxic area (Fig. 2e) for the Baltic Proper were discussed above.

In the Northern Adriatic, summer bottom dissolved oxygen decreased from the 1960s until the early 1990s; the most recent data may suggest an increase in dissolved oxygen, but this is inconclusive as data are sparse (Fig. 4f).

The spatial extent of the biologically diverse community dominated by the red alga *Phyllophora* spp. was already contracting in the Black Sea by the time bottom hypoxia became a regular occurrence in the early 1970s (Fig. 5g) (Zaitsev and Mamaev, 1997;

Langmead et al., 2009). The hypoxic area increased throughout the 1970s and 1980s, peaking in 1990 (Fig. 5f). Faunal mortalities were common during the 1970s and 1980s (Bodeanu et al., 2004). These decades also saw a decrease in biomass of the bivalve *M. galloprovincialis* (Fig. 5h) which succumbed to prolonged periods of hypoxia (Zaitsev, 1998; Langmead et al., 2009); however, this trend may also be partially explained by predation by the invasive carnivorous sea snail, *Rapana venosa* (Zaitsev, 1992). As phytoplankton biomass decreased during the 1990s (Fig. 5a), the hypoxic area also decreased (Fig. 5f). Recent evidence suggests that *Phyllophora* beds are beginning to recover (Mee et al., 2005). The changes in hypoxic area are likely related to nutrient inputs from the Danube River and associated pelagic productivity; the reduced hypoxic area during the 1990s corresponds well with a decrease in nutrient concentrations as a result of the major reduction in industrial farming operations in the catchment (Mee et al., 2005).

4. Discussion

4.1. Comparison of study areas

Europe's regional seas clearly show conventional (i.e. Pearson and Rosenberg, 1978; Cloern, 2001) responses to nutrient enrichment such as enhanced phytoplankton growth and organic loading of benthic environments. However the precise effects, as well as the course and recovery from eutrophication following reduced nutrient loading, vary widely. Cyanobacterial blooms are a feature only of the Baltic with its low salinities (Wasmund, 1997). Proxy indicators of these blooms suggest that eutrophication is worsening in this system. The coastal North Sea shows a clear increase in phytoplankton biomass. While in some systems increased phytoplankton biomass may correspond to an increase in nutrients, in the North Sea TN and TP are decreasing in coastal waters and phytoplankton are probably responding to other environmental factors (McQuatters-Gollop et al., 2007). Although declining, nutrient concentrations remain in excess indicating that eutrophication continues to be a concern in some near-shore areas (Lancelot et al., 2006; Carstensen et al., 2007). A probable explanation is that phytoplankton in coastal North Sea waters have previously been light-limited (Pätsch and Radach, 1997), but the present hydrological regime has reduced light-limitation allowing them to better use available nutrients (McQuatters-Gollop et al., 2007).

Three regional seas, and perhaps also the Baltic (see Jaanus et al., 2007) show an increasing trend in the proportion of diatoms relative to dinoflagellates. The causes of this trend need further investigation. CPR data for the coastal North Sea show an increase in phytoplankton biomass but a decrease in phytoplankton abundance. This could be explained by changes in the nanoplankton community, as reported by Hickel (1998) for Helgoland Roads, or in phytoplankton cell size, as reported by Philippart et al. (2000) for the Wadden Sea. However, this phenomenon has been observed throughout much of the North Atlantic (Leterme et al., 2006) and so is unlikely to be caused by eutrophication. Finally, changes in Si availability due to increased water temperatures may provide a partial explanation for the increase in diatom abundance (Natori et al., 2006).

Effects on benthic ecosystems have been difficult to ascertain consistently over the study areas, but the Black and Baltic Seas have both suffered from hypoxia aggravated by eutrophication. However, in the Baltic Proper, hypoxia is closely interwoven with natural climate events that drive saline intrusion as well as with eutrophication. In the Northern Adriatic and northwest Black Sea, indicators show a decline in phytoplankton biomass and an improvement in benthic oxygen conditions (although change in

bottom oxygen is much less clear in the Northern Adriatic) suggesting that these two ecosystems are recovering from eutrophication. While reduced nutrient loading (Artioli et al., 2008) is the most likely reason for recovery of both ecosystems, it is the result of abatement for the Northern Adriatic but of economic collapse for the Black Sea shelf. With the dissolution of the Soviet Union and collapse of industrial farming in the shelf's catchment, as well as implementation of EU policy directives (notably the Nitrates and Urban Waste Water Treatment Directives) in Western Europe, nutrient loads to the Black Sea shelf declined and nutrient concentrations in the water column reflect this. The change in N:P ratio may be explained by more effective P relative to N abatement in the catchment. However, a reversal in trends could occur on the Black Sea shelf if economic recovery is not paired with nutrient management.

A key factor distinguishing the four basins is exchange and residence time. Here the Baltic Proper stands out as a basin particularly sensitive to nutrient inputs. Limited exchange and long retention times mean that current eutrophication is a legacy of past nutrient loads. The Black Sea, which also has limited exchange, has a sink that removes nutrients from biological availability. Once nutrients have passed through the halocline via phytoplankton sedimentation to the vast and deeper central water body of the Black Sea, they are removed for time scales of hundreds of years (Sorokin, 2002). The shallower Baltic Sea also has a halocline capturing nutrients below. However, due to its shallowness, sediment–water interactions and varying oxygenation status induced by vertical fluctuations of the halocline lead to frequent exchange of P to and from the sediments. During times when the halocline becomes shallower and larger areas become anoxic, annual P mobilization may correspond to some 20% of total P stocks in the Baltic proper (Conley et al., 2002). Thus, both seas have large internal nutrient pools below the halocline, but in the Baltic case this pool is a temporary one that is easily remobilised when conditions change. Changes in the vast deep nutrient pool of the central Black Sea are much less influenced by anthropogenic emissions, at least on decadal time scales. It is possible that this sink has spared the Black Sea from worse eutrophication impacts, but better information about exchange within the Black Sea is clearly needed (see also Artioli et al., 2008). The coastal North Sea lies at the other extreme, being well flushed by the Atlantic Ocean. Relative to the Baltic and Black Seas, eutrophication in the coastal North Sea has had relatively mild and localized effects despite substantially heavier nutrient loading (EEA, 2005; Artioli et al., 2008; Vermaat et al., 2008). The Northern Adriatic is intermediate in terms of exchange, and also in terms of the severity of eutrophication.

4.2. Climate confounds the eutrophication signal

European marine ecosystems are subject to a range of anthropogenic pressures such as chemical contamination, habitat loss, and overharvesting. Any suite of indicators for eutrophication needs to be embedded within a larger indicator suite capturing a wide range of interrelated or synergistic anthropogenic pressures as well as natural variability. Climate deserves specific mention as the discussion has shown that it can be difficult to separate the effects of climate variability from those of eutrophication. Either, or both, may lie behind increased algal productivity (McQuatters-Gollop et al., 2007), altered food webs (Daskalov, 2003; Österblom et al., 2007), changes in phytoplankton community composition with increases in undesirable or harmful species (Sellner et al., 2003) and bottom hypoxia (Conley et al., 2002).

A climate-driven regime shift in the North Sea during the 1980s (Beaugrand, 2004), with effects on water transparency and

phytoplankton biomass, could explain inconsistencies in regional indicator trends, notably increased chlorophyll coinciding with increased Secchi depth (McQuatters-Gollop et al., 2007). The North Sea has shifted from a boreal system to a warm temperate system, and its characteristic features have changed, with North Sea waters becoming increasingly warm and clear. Consequently, increases in abundance of warm water zooplankton (Beaugrand et al., 2002) and phenological changes have also been recorded in the region (Edwards and Richardson, 2004). The 1980s regime shift was not limited to the North Sea/North Atlantic. Near-synchronous ecological shifts also occurred in the Mediterranean (Molinero et al., 2005; Conversi et al., in press), Pacific (Hare and Mantua, 2000), Baltic (Alheit et al., 2005) and Black Seas (Niermann et al., 1999; Oguz and Gilbert, 2007), further supporting the likely climate-driven nature of these changes.

In the Baltic Sea, the extent of bottom hypoxia is interwoven with climate-modulated saline incursions and exchange across the halocline (Conley et al., 2002), as well as organic loading from an enriched pelagic zone. Nutrient budgets indicate that P reflux from sediments is by far the predominant source of P (Savchuk, 2005; Artioli et al., 2008), emphasizing both the role of climate and the legacy of past nutrient enrichment on current ecosystem states. However, no clear trends are observable in cyanobacteria (for the reasons mentioned in Section 2.2) or *in situ* chlorophyll concentrations (Fig. 2a). There is also evidence of a regime shift in the Baltic pelagic ecosystem, but the relative roles of climate, eutrophication and fishing pressure are still under debate (Alheit et al., 2005; Österblom et al., 2007; Möllmann et al., 2008).

It is postulated that climate also plays a role on the northwest Black Sea shelf, with mild winters suppressing vertical mixing and limiting phytoplankton blooms (Oguz, 2005). However, this relationship appears to be non-linear; in 2001 a climate anomaly, with unusually high sea surface temperatures and increased stratification, caused sea-wide algal blooms, an increased dinoflagellate fraction and bottom hypoxia, all of which have been previously associated with eutrophication (McQuatters-Gollop et al., 2008).

The observed decline in oxygen concentrations in the Northern Adriatic during the 1970s and 1980s may not be a consequence of eutrophication as this indicator has previously been found to be modulated primarily by climate rather than nutrient loading in the Northern Adriatic (Degobbis et al., 2000). If so, this could explain the decline in bottom oxygen despite the decreasing impact of eutrophication on the pelagic indicators mentioned above.

4.3. Insights provided by ecosystem indicators

Phytoplankton biomass is a common indicator of eutrophication. The trends indicated in Figs. 2–5 follow both general and site-specific knowledge of the effects that eutrophication has on pelagic ecosystems, and so this indicator provides consistent insights. Unfortunately similar measurements are not undertaken in different seas, so comparison across seas is limited. This will be a major upcoming challenge for the European Marine Strategy Directive, which requires indicators and their assessment approaches and methods to be comparable both within and between marine regions (European Commission, 2008). The Baltic Sea poses particular difficulties for this indicator as due to the problems with direct measurement Secchi depth is used as a proxy for phytoplankton biomass; however Erlandsson and Stigebrandt (2006) show that it is possible to accommodate interference. The use and interpretation of Secchi depth in the coastal North Sea also pose particular problems. Here, Secchi depth cannot be used as a proxy for phytoplankton biomass. In the Northern Adriatic and northwest Black Sea shelf trends in Secchi depth do appear to follow those of phytoplankton biomass. However, use of Secchi

depth to proxy phytoplankton biomass requires good understanding of the underlying mechanics and sources of interference. Site-specific features mean that it cannot be used on its own to compare phytoplankton biomass across regional seas.

Unfortunately, the N:P ratio does not provide consistent insights into phytoplankton composition. There is no recurrent pattern across the four study areas. In the Black Sea, N:P increases coincide with a greater proportion of diatoms in the phytoplankton community. A similar trend might be occurring in the coastal North Sea since 1995. However in the Northern Adriatic, the diatom fraction increased as the N:P ratio declined. A general increase in diatom fraction is probably attributable to other factors, notably the supply of silicate. Consequently, the N:P ratio is a poor ecosystem indicator and will remain so until a clear relationship with other ecosystem variables can be verified. Nevertheless, we identify two instances where it is useful: it has potential for the Baltic where declining N:P ratios could augment Secchi depth to indicate increased cyanobacterial activity; and the N:P ratio can be used to identify the most probable growth-limiting nutrient, essential information for nutrient management strategies. The increase in diatom fraction in all seas could suggest 'good news', particularly if an increase in fodder zooplankton results (Verity et al., 2002). The use of compositional changes in the phytoplankton community as an eutrophication indicator clearly needs further work. Continued monitoring of phytoplankton community compositions will be needed to elucidate any relationship with nutrient ratios.

In some systems bottom hypoxia may offer good insights into responses to eutrophication, although care should be taken with interpretation as its link to eutrophication varies regionally. For the northwest Black Sea shelf bottom hypoxia is clearly a product of excessive phytoplankton growth. For the Baltic, it is the consequence and cause of excessive phytoplankton growth because of its effect on sedimentary reflux of P. Along with Secchi depth and N:P ratio, it is an indicator of mechanisms that drive cyanobacterial growth. Separation of eutrophication and climate signals in bottom hypoxia remains a challenge for further research: for instance, bottom dissolved oxygen in the Northern Adriatic has not responded solely to changes in phytoplankton biomass, but is also largely regulated by climatic events (Degobbis et al., 2000).

The response of benthic biological indicators to eutrophication is somewhat unclear. Depth limitation of seagrasses, such as *Zostera*, in the Baltic is known to be strongly influenced by increased turbidity following nutrient enrichment, and has been proposed as a bioindicator under the European Water Framework Directive (Krause-Jensen et al., 2005; Schories et al., 2009). Depth limitation of Baltic macroalgae, such as *Fucus vesiculosus*, appears to function less straightforwardly as an indicator, possibly due to a more complicated response to eutrophication (Torn et al., 2006). In the Black Sea, however, the decline and possible recovery of the macroalga *Phyllophora* appear to be closely linked to eutrophication (Mee et al., 2005). It has previously been observed that responses to increased nutrients vary across macroalgal taxa (Fox et al., 2008) and, while they were not explored during ELME, other macroalgal groups, particularly filamentous opportunistic genera such as *Ulva*, *Enteromorpha*, *Ectocarpus* and *Polysiphonia*, have also been used successfully as eutrophication indicators in European seas (Korpinen et al., 2007; Scanlan et al., 2007). This may mean that opportunistic groups are more useful macrophyte indicators of eutrophication than as perennial macroalgae. Causes for trends in zoobenthic biomass are also unclear. It is possible that changes observed in the macrozoobenthos of the northwest Black Sea shelf reflected an initial increase in biomass with the onset of eutrophication followed by a decrease as the shelf hypoxic area expanded as implied by Mee et al. (2005), but the data are insufficient for inferring conclusive trends. As in the

Baltic Proper, trends in Black Sea macrozoobenthos biomass can be attributed to a range of factors and so this indicator provides only limited insights into the effects of eutrophication on benthic ecosystems.

One reason for the lack of clear relationships between time-series could be due to the spatially inhomogeneous nature of the datasets. Many of the time-series used here are based on repeated measurements at one station or in a particular localized area. Therefore in some of our study regions spatial discrepancies exist between datasets, making the identification of relationships especially difficult. Even in regions where some spatially comprehensive data do exist time-series of other indicators at an equivalent scale are rarely available; for example, the Continuous Plankton Recorder survey provides plankton data for much of the North-East Atlantic, but other ecosystem indicators are largely restricted to more limited spatial areas (e.g. nutrient and Secchi depth data are primarily available only for the North Sea and a few other coastal regions). This mismatch of scale is a real problem in marine research which can only partially be solved by modelling exercises which extrapolate information to a larger spatial scale; however, there is no substitute for thorough monitoring programs.

4.4. Implications for policy

This paper has focused on ecosystem state indicators for which long time-series were available, although coverage has not always been complete or consistent. Our requirement for a long time-series aimed at capturing ecosystem states prior to the emergence of eutrophication symptoms, the period when eutrophication was at its worst, and contemporary times. The last period is characterised by effort to abate nutrient emissions and loads to regional seas. Policies addressing eutrophication have been implemented throughout much of the European Union, in other words most countries located in the catchment of Europe's regional seas. But the EU Water Framework Directive remains at an early stage of implementation and there are new pressures to expand agriculture for fuel crops as well as food and fibre. Our analysis highlights a very obvious implication for policy: that regional specificity in ecosystem response to anthropogenic nutrient loading means that policies must be tailored to each regional sea. Regional variation in policy is in accordance with the new Marine Strategy Directive; however the societal acceptability in different regions that implement abatement measures of varying severity will be an issue for the political arena. We identify three aspects from the discussion above that can be expected to influence regionalised policy development and implementation: (1) the severity of eutrophication, past and present; (2) evidence of ecosystem recovery; and (3) other sources of pressure on marine ecosystems.

The severity of eutrophication is coupled to the urgency for policy development. The Baltic Proper, with its positive feedback linking pelagic and benthic ecosystems, stands out as the regional sea with the worst symptoms of eutrophication and with the greatest urgency for policy action. Increased phytoplankton biomass in the coastal North Sea also suggests a degree of urgency. Climate is an important regulator of phytoplankton biomass across the North Sea and climatic changes have provided favourable conditions for phytoplankton growth. Although nutrient concentrations are declining in the coastal North Sea, nutrients remain in excess and coastal phytoplankton are not nutrient limited (McQuatters-Gollop et al., 2007; Artioli et al., 2008; Vermaat et al., 2008). For the Northern Adriatic and northwestern Black Sea shelf, current ecosystem conditions appear to be improving. However, while recovery in the Northern Adriatic can be attributed to policy and nutrient abatement, ecosystem recovery in the Black Sea has not occurred due to policy effectiveness, but is partly a product of economic collapse. Concerns about continuation of current trends in the Black Sea, and the possibility of their reversal should economic recovery not be paired with nutrient controls, highlight urgency for policy development and implementation in the catchment of the northwest Black Sea shelf.

The indicators do not suggest recovery in the Baltic despite nearly two decades of policy implementation and economic collapse in former communist countries. An important eutrophication symptom, cyanobacterial blooms are fuelled by sedimentary nutrient recycling and a positive feedback that augments P availability and may imply that the system can be expected to be relatively insensitive to policy measures. A scenario analysis (Wulff et al., 2007) has shown that policy implementation could lead to recovery over a time frame of several decades (Savchuk and Wulff, in press). Thus the commonly felt urgency (HELCOM, 2006) of eutrophication abatement measures may lose its momentum and social acceptance given that a slow system response will not reveal any improvement during a decade or more. Recent studies by Casini et al. (2008) and Daskalov (2002) suggest that top-down control, i.e. the disappearance of top predators, has a cascading effect all through the food web, eventually resulting in an increase in phytoplankton biomass as grazer levels are reduced.

The third aspect we identify relates to the multiple sources of pressure on marine ecosystems, such as the confounding effects of fisheries (Daskalov, 2002) and climate (McQuatters-Gollop et al., 2007; Artioli et al., 2008; Vermaat et al., 2008). Understanding the natural variability of fish and plankton populations through monitoring is a requirement of the Marine Strategy Directive which may eventually enable the separation of eutrophication effects from those of other ecosystem pressures. In particular nutrient abatement and system response must be carefully considered in the context of potentially strong climatic influences. Proper consideration of climate introduces challenges for policy development and uncertainties as to when and if expected benefits will materialise.

5. Conclusions and recommendations

Eutrophication impacts and ecosystem responses are spatially variable across Europe's regional seas. Due to differences in indicators monitored and assessment methodologies used it is difficult to compare eutrophication status among seas. Regionally and methodologically consistent monitoring programmes are needed in order to facilitate interregional comparability; this is a requirement of the Marine Strategy Directive.

Further insight into the causes of site-to-site variability in ecosystem response to nutrient loading, along with comprehensive monitoring data, may aid the development of a general predictive framework accounting for nutrient effects in conjunction with the moderating influences of local physical, chemical and biological factors. Predictive models are essential for the management and control of coastal marine eutrophication, and continued research into their development is required. Comparative analyses such as this one will be necessary in order to determine the degree to which local eutrophication indicators are (or are not) directly and causally linked to changes in nutrient loading to the systems being studied. Additionally, standardization of methods used and indicators monitored between regional seas will make such analyses more robust. However, this need must be balanced with the continuation of established monitoring programmes which are providing crucial long-term time-series of data, a point emphasised in the Marine Strategy Directive.

The indicators suggested in the Marine Strategy Directive have the potential to provide scientifically founded information

regarding eutrophication assessment to policy-makers – but only if multiple indicators are employed as a suite. Due to regional variability in ecosystem response to increased nutrients, no sole indicator, not even nutrient concentrations themselves, can offer a comprehensive and comparable insight into the eutrophication status of European seas. To assess eutrophication effectively, indicators suites must be supplemented with sea-specific knowledge of ecosystems characteristics and processes; lists of indicators are not sufficient to determine good environmental status without an adequate understanding of each system's attributes gained through scientific research. Indicators must also be assessed on similar spatial scales in each regional sea. The interaction of multiple pressures remains a key challenge for assessing eutrophication as do separating changes in ecosystem parameters resulting from nutrient loading from those occurring due to natural seasonal and interannual dynamics. The continuation of established monitoring programmes (e.g. the Continuous Plankton Recorder survey (Batten et al., 2003; Brander et al., 2003, among others), the Romanian National Institute for Marine Research and Development time-series (Bodeanu et al., 2004; Cociasu and Popa, 2004, among others), the Helgoland Roads time-series (Wiltshire and Manly, 2004; Wirtz and Wiltshire, 2005, among others), and the Dutch Noordwijk and Terschelling monitoring transects (de Vries et al., 1998; Los and Wijsman, 2007, among others)) will extend existing datasets and provide baselines with which to compare future indicator trends; the Marine Strategy Directive specifically requests that existing monitoring operations should form the basis of future programmes in order to avoid duplication of effort. Coherence of monitored indicators and methodologies will facilitate comparison of eutrophication effects between European seas.

Acknowledgements

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