Testing of indicators for the marine and coastal environment in Europe

Part 3: Present state and development of indicators for eutrophication, hazardous substances, oil and ecological quality

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Summary

This report describes the present state and development of policy relevant indicators for eutrophication, hazardous substances, oil and ecological quality in the marine and coastal environment in Europe related to the input of substances affecting these issues. The indicators tested and developed here are part of the EEA core set of water indicators. *Assessments on state and trends are given in italics* to distinguish them from information on indicator development.

The analysis in this report refers to the DPSIR assessment framework. **D**riving forces (or human activities) lead to **P**ressures (emissions of nutrients and hazardous substances) on the environment. As a result, changes in the **S**tate of the environment may lead to **I**mpacts on ecosystems and human health and societal **R**esponses must be defined to reduce the adverse effects. The focus of the report is on pressure, state and impact indicators, taking into consideration the European Union's policies to reduce eutrophication and pollution in the context of its strategies towards biodiversity and sustainable development.

Not all organisations including Marine Conventions and Action Plans accept the indicator approach, although their data sets are being used for the testing of indicators. Therefore the development status of an indicator based on existing data sets is classified in this report by the indicator potential of datasets and the progress made in development of an indicator. A high indicator potential is given to variables that have communicative power (easy to understand) and statistical power (trends may be detected easily). Presenting results for these variables should preferably be done in illustrations (figures, graphs or maps) showing the trend from year to year or in three to five year periods, depending on the frequency of monitoring and assessment. For updating indicators annually, the time series may be extended easily and results remain comparable. None of the indicators are classified as completed in development, meaning that a lot of work still has to be done.

Excess supply of nutrients from human activities into water bodies enhances primary production of algae. This process is defined as eutrophication. Eutrophication effects are observed in many European coastal waters. Eutrophication indicators with high potential are: inputs of nutrients into marine and coastal waters (pressure indicator) and nutrient concentrations in coastal waters (state indicators). Oxygen and chlorophyll a are chemical impact indicators under development.

Hazardous substances are toxic, persistent and/or liable to bio-accumulate. The heavy metals cadmium, mercury, lead and zinc, the persistent organic pollutants lindane and PCB_7, and mineral oil were selected for indicator development. Indicators for hazardous substances with high potential are: direct and riverine inputs (pressure), concentrations of hazardous substances in the blue mussel and oil slicks observed (both state), and some biological effects of hazardous substances (impact). With respect to the latter a selection among the various biological effect techniques, such as imposex in marine snails, should be made of biological effects that can be clearly related to hazardous substances as cause of the effect and that offer possibilities of attracting the attention of policy-makers and the public.

Integrated indicators describe the ecological quality of marine and coastal waters, which might be impacted by different pressures (multi-stress). These integrated indicators (soft-bottom benthos and phytoplankton/ phytotoxins) are at an initial stage of development. Hence, knowledge of, and experience with, the use of these indicators is limited and restricted to the local and regional level.

The quality assurance and completeness of the data sets used for trend analysis should get highest attention by data providers. The value of data used for trend analysis increases considerably when the data are reliable and complete. The temporal and spatial coverage of data collection should be increased to include all European regional seas and coastal waters and to avoid gaps in the time series for a reliable, comprehensive assessment and geographically and temporally balanced indicators.

More research is recommended to separate the natural and anthropogenic loads of nutrients and hazardous substances in rivers, as well as on natural variations in these loads related to variations in river flow. A better understanding of these parameters would make it possible to determine input loads due to human activities only and to correct riverine loads for natural variations caused by rainfall and river flow. This will improve the value of the input indicator considerably.

More research is recommended on the natural state of marine and coastal waters related to the presence of nutrients and hazardous substances depending on local conditions and natural variations. By comparing the actual state of the marine and coastal waters with the natural one, marine and coastal waters may be classified, making it possible to compare the different marine and coastal waters in an easy and standardised way.

Scientists may be of special help to e.g. EEA, Marine Conventions, DG Environment in clarifying cause–effect relations within the DPSIR assessment framework. They can advise on the development of information and monitoring strategies. To achieve the goals of effectiveness and efficiency of monitoring, scientific help is really indispensable. For instance, the efforts put into: (a) model development, (b) direct measurements, and (c) remote sensing may be combined and attuned to each other. By combining results of the three data sources, an optimal strategy may be developed using the specific strengths of each source. As a result, spatial and temporal coverage may improve, information may become more reliable and efforts put into generating the information needed may decrease.

1. Introduction

1.1. Background

In the report by the European Environment Agency (EEA) *Environment in the European Union at the turn of the century* (EEA, 1999a) the actual and foreseeable state of the environment in EU and accession countries is assessed. The outlook was based on socioeconomic and environmental policies that are assumed to be implemented by 2010. It describes the interrelations between human activities and the environment; it serves to inform policy-makers on developments in environmental parameters and the effects of measures taken. As such the report is a background for strategic policy development. The analysis presented in this report follows the DPSIR assessment framework. It starts with **D**riving forces (or human activities), which lead to **P**ressures (emissions) on the environment. As a result, changes in the **S**tate of the environment may lead to **I**mpacts, and **R**esponses must be defined to reduce the adverse effects.

In a subsequent EEA report *Environmental signals 2000* (EEA, 2000a), the indicator assessment based on the DPSIR framework was worked out more quantitatively, presenting among others eutrophication indicators for coastal waters. Subsequent EEA *Environmental signals* reports provide a yearly overview on selected policy themes.

In the EEA report *Europe's environment: The second assessment* (EEA, 1998) main subjects identified as themes for concern in the marine and coastal environment chapter are: eutrophication, contamination (later addressed as hazardous substances), overfishing and degradation of coastal zones. The EEA report *Environment in the European Union at the turn of the century* (EEA, 1999a) discusses the main challenges and problems in the coastal zones of the four regional seas (the North-East Atlantic, the North Sea, the Baltic, the Western Mediterranean). The pressure of economic growth and spatial development differs between the regions. Due to various ecological qualities, the coastal zones of the regional seas have a different sensitivity to the pressures on the coastal zone.

Other recent indicator reports also describing indicators for the marine and coastal environment are the Eurostat report *Towards environmental pressure indicators for the EU* (Eurostat, 1999) and the Baltic 21 report *Developments in the Baltic Sea region towards the Baltic 21 goals — An indicator-based assessment* (Baltic 21, 2000). These reports have their own communication aims and policy context and can be considered as examples of comprehensive indicator reporting.

The present report follows the DPSIR assessment framework and describes the present state and development of indicators related to the input of substances into the European marine and coastal waters due to human activities. Other themes of concern such as overfishing and degradation of coastal zones are not addressed in this report. This report has made use of data from the marine conventions and EEA member countries brought together by the Topic Centre Marine and Coastal Environment of the EEA in 1998 and 1999.

1.2. Scope

The report evaluates the potential to provide indicators from best available data and knowledge and not particularly from the context of best needed data. Best needed data considering eutrophication, hazardous substances and oil at the European level are presently being identified through the implementation of the Water Framework Directive (WFD) and discussed in international working groups. No effort has been made to use statistical trend detection methods or to add to ongoing scientific work within the marine conventions and the International Council for the Exploration of the Sea (ICES). The time series analysis has been presented graphically using annual data, taking into account estimates where appropriate. However the report reflects on the need to envisage the statistical power of (future) data sets and indicators in relation to the wanted accuracy level to assess changes over a certain period of time for a particular region.

Chapter 2 deals with eutrophication. Using the DPSIR framework the current state with respect to indicator development for this theme is described. Pressure, state and impact variables are evaluated on their indicator potential and trends are analysed.

Chapter 3 is dedicated to hazardous substances (heavy metals and persistent organic pollutants (POPs)) and Chapter 4 to oil spills. These chapters describe the present state of indicator development for these substances. Again pressure, state and impact variables are evaluated on their indicator potential.

In Chapter 5 attention is given to biological quality variables that may be used as integrated indicators to present multi-stress effects due to various disturbances. The development of biological quality variables is still in a preliminary phase.

Chapter 6 discusses the state of the art with respect to indicator development for the marine and coastal environment. Attention is given to the indicator potential of the different parameters that may be used as indicators and the progress made in development. This chapter also addresses the need for, and possibilities of, updating and upgrading the indicators and to adjusting them to changing settings (time and geographical coverage, integration over substances and species).

Chapter 7 summarises the conclusions and recommendations on the state of the art of indicator development and on trends analysed concerning the state of the marine and coastal environment.

Annex 1 contains the testing report of two eutrophication parameters, oxygen and chlorophyll, applied to time series of individual stations in Danish waters. The main findings of this contribution to the testing have been integrated in Chapter 2. In view of other ongoing indicator developments within the conventions and EEA member countries and in light of the coastal and marine indicator workshop organised in June 2001 by the EEA in collaboration with the conventions and member countries, the full report is included.

Annex 2 presents the testing report of possible indicators for soft-bottom zoobenthos based on time series from Greek waters. The main findings of this testing have been included in Chapter 5.

Annex 3 shows the detailed testing results of an indicator on phytotoxins. The main findings of this indicator have been included in Chapter 5.

1.3. European marine and coastal waters

Three large marine regional seas were initially distinguished by the EEA prior to addressing the wider area of Europe (see Figure 1):

- the Helcom area: the Baltic Sea, where it borders EEA countries;
- the North-East Atlantic including the North Sea (OSPAR area): the Norwegian Sea, the North Sea, the Celtic seas, the Bay of Biscay and the Iberian coast;
- the UNEP/MAP area: the Mediterranean Sea, where it borders EEA countries.

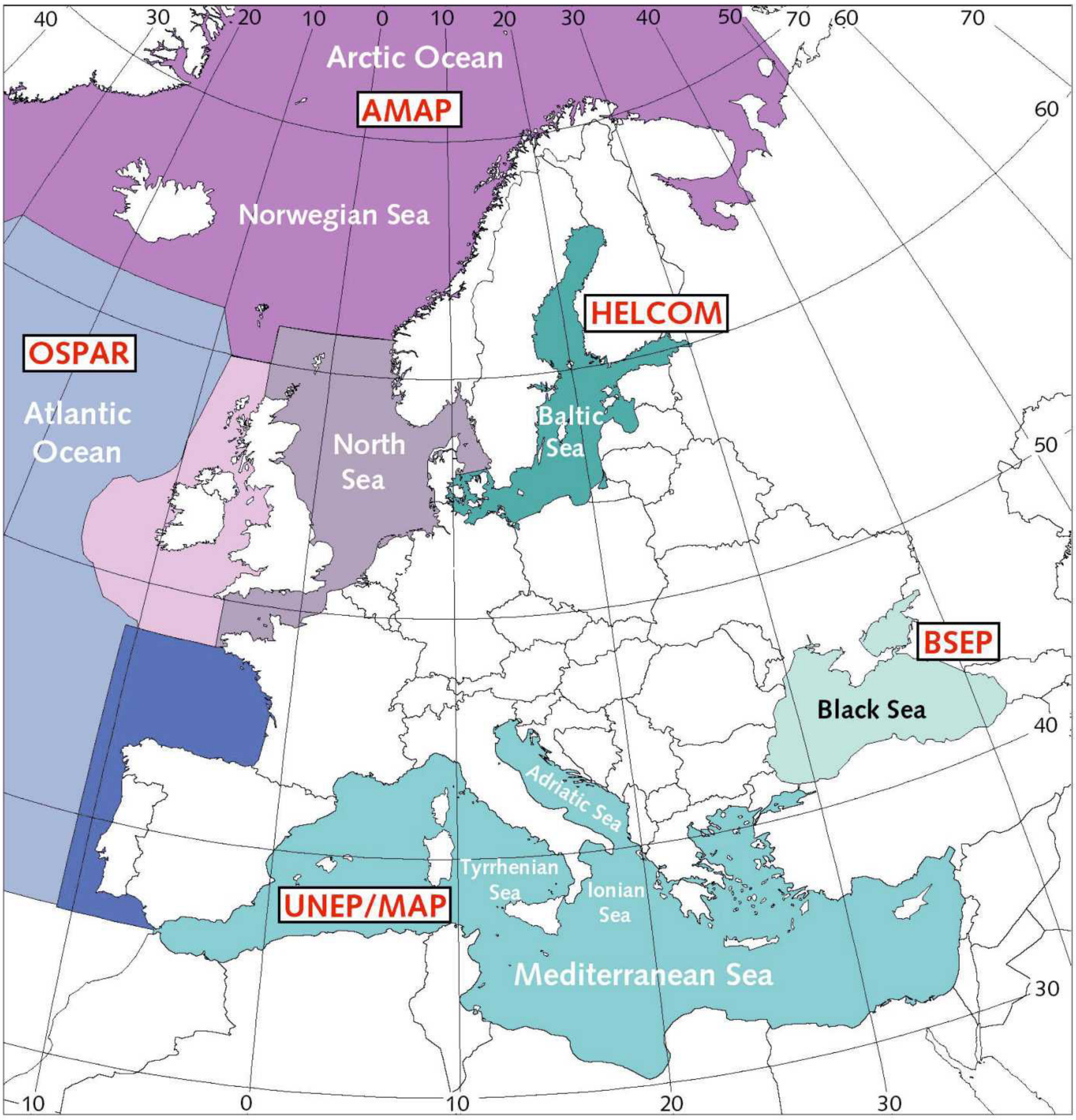
The Black Sea will be included in future pan-European reporting following the enlargement of the EEA towards the east.

The Baltic Sea is the second largest brackish water area in the world and consists of several sub-basins separated by sills. Due to the low salinity the biodiversity is lower than in both more saline waters and in freshwaters, and many species live at the edge of their ability. Large-scale mixing of Baltic surface water and saline North Sea water takes place in the Belt Sea and Kattegat (Helcom, 1996).

The water mass in the Norwegian Sea is a mixture of low saline Baltic water, North Sea water and Atlantic water mixed in the Kattegat and Skagerrak. The North Atlantic current brings warm and high saline water masses to the Norwegian Sea. In addition to this saline water mass there are a number of different water masses found in fjords and other estuaries along the coastline. Wide ranges of water characteristics are found, depending on estuary type, residence time, freshwater input, mixing conditions, etc.

The North Sea is often divided into seven sub-areas: the southern North Sea, the central North Sea, the deeper northern North Sea, the Norwegian Trench, Skagerrak and finally the shallow Kattegat as a transition zone to the Baltic Sea, and the Channel as a transition zone to the North-East Atlantic. The shallow southern North Sea includes the Wadden Sea tidal area, the German Bight and the Southern Bight. The largest freshwater source to the North Sea is the

Figure 1 The European regional seas



Baltic Sea. Most of the Atlantic inflow circulates in the deeper part of the northern North Sea, the Norwegian Trench and Skagerrak, before it enters the Norwegian Sea. A small proportion of water enters the North Sea through the English Channel. The flushing time for the entire North Sea is estimated to be in the range of between 365 and 500 days. In periods with westerly wind, low saline water, carrying nutrients and hazardous substance from the rivers in the southern North Sea, is carried up the Jutland west coast to the Skagerrak (OSPAR/QSR, 2000).

The Celtic seas include the area located south of Ireland, Saint George's Channel, the Irish Sea and North Channel between Ireland, Scotland, England and Wales and finally the shelf areas west of Ireland and Scotland. Bands of less saline water are found around Ireland and Scotland due to run-off of freshwater. It is estimated that the flushing time is one to two years in the Irish Sea and shorter in the more open areas. In addition to the water masses in the open areas there are a number of different waters found in fjords and other estuaries along the west coast of Scotland and the Scottish islands. Wide ranges of water characteristics are found, depending on estuary type, residence time, freshwater input, mixing conditions, etc.

Most of the surface waters found in the Bay of Biscay and along the Iberian coast have an Atlantic origin. Deeper water masses may also be a mixture of Atlantic and Mediterranean water masses. Coastal upwelling is a dominant process in summertime off the Iberian west coast and in the south-western part of the Bay of Biscay. Upwelling takes place in rather narrow bands, so-called filaments, along the coast (OSPAR/QSR, 2000, Region IV, 2000).

In the Mediterranean Sea evaporation exceeds precipitation and freshwater load. As a result there is a net inflow of water from the Atlantic Ocean and the overall effect is a very high salinity in the Mediterranean Sea, increasing from west to east. The flushing times are long, some 80 to perhaps 300 years (UNEP/MAP, 1996). The Mediterranean is divided into two basins separated by the Sicilian Channel. Tidal amplitudes are small in the Mediterranean Sea.

1.4. Objectives

The development of a common set of indicators was stressed at the Fourth European Conference of Ministers for the Environment in Aarhus in June 1998. Indicators can play a vital part in focusing and illuminating the significance of environmental change and the progress to sustainable development. Indicators are quantified information that help to explain how the quality of the environment changes over time or varies spatially.

Providing reliable and relevant data and information to support widely agreed key indicator sets in a consistent and timely way should be one main objective of the improvement of monitoring and data gathering. To achieve this, the EEA works as facilitator between member countries, Community institutions and other environmental organisations and programmes.

The aim of this report is to provide an insight into the progress made in developing indicators that describe the interrelations between human activities and the marine and coastal environment as related to the inputs of substances.

1.5. Inputs of substances, concentrations in coastal waters and effects on marine ecosystems

Depending on the nature of substances and their fate in the environment the input of substances into marine and coastal waters influences the marine and coastal water environment in a different way. With respect to ecological risks involved a distinction is made between nutrients and hazardous substances.

The ultimate aim of the Water Framework Directive is to achieve the elimination of priority hazardous substances and contribute to achieving concentrations in the marine environment near background values for naturally occurring substances (EU, 2000).

Nutrients (nitrogen and phosphorus compounds) are present naturally in the marine and coastal water environment and are essential to support life (ecosystems' food chains). Ærtebjerg et al. (2001) define eutrophication as enhanced primary production due to excess supply of nutrients from human activities, independent of the natural productivity level for the area in question. In severe cases of eutrophication, dead algae use up the oxygen, bottom dwelling animals die, fish either die or escape and significant overall ecological damage make the quality unacceptable. Eutrophication also increases the risk of blooms of toxic phytoplankton species, which may cause death of fish and benthic fauna, and poison humans.

Hazardous substances are defined as substances or groups of substances that are toxic, persistent and liable to bio-accumulate and other substances or groups of substances that give rise to an equivalent level of concern (EU, 2000). Hazardous substances may be divided into heavy metals, persistent organic pollutants (POPs) and mineral oil.

Heavy metals are metals or metalloids which are stable and which have a density greater than 4.5 g/cm^3, namely lead, copper, nickel, cadmium, platinum, zinc, mercury and arsenic (EEA, 1999a).

POPs are chemical substances that persist in the environment, bio-accumulate through the food web, and pose a risk of causing adverse effects on human health and the environment (UNEP, 1998; UNECE, 1998).

The fate and ecosystem effects of mineral oil in marine and coastal waters differ from those of other hazardous substances. Generally, oil slicks occur as the result of intentional or accidental discharges of oil from ships or offshore installations. The most conspicuous effects are the contamination of sea birds in coastal as well as offshore areas. Indicators for mineral oil are dealt with separately, with a focus on oil slicks.

2. Eutrophication indicators

2.1. Introduction

2.1.1. Policy questions

As defined by Ærtebjerg et al. (2001) eutrophication is enhanced primary production due to excess supply of nutrients from human activities, independent of the natural productivity level for the area in question. Other definitions are in use as well, all stressing the enrichment of water by nutrients and the accelerated growth of algae. For instance the Urban Wastewater Treatment Directive 91/271/EEC defines eutrophication as the 'enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, 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'.

Eutrophication is of major concern in some parts of the European seas (EEA, 1998).

In order to reduce the adverse effects of a surplus of anthropogenic input of nutrients and to protect the marine environment, measures are being taken as a result of various initiatives at all levels (regional conventions and ministerial conferences, and at a European, national and global level): the UN global programme of action for the protection of the marine environment against land-based activities; the Mediterranean action plan (MAP), adopted in 1975; the 1992 Helsinki Convention on the Protection of the Marine Environment of the Baltic Sea Area; the 1998 OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic. Conventions acknowledge the importance of compliance with EC directives such as Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources, and corresponding national legislation.

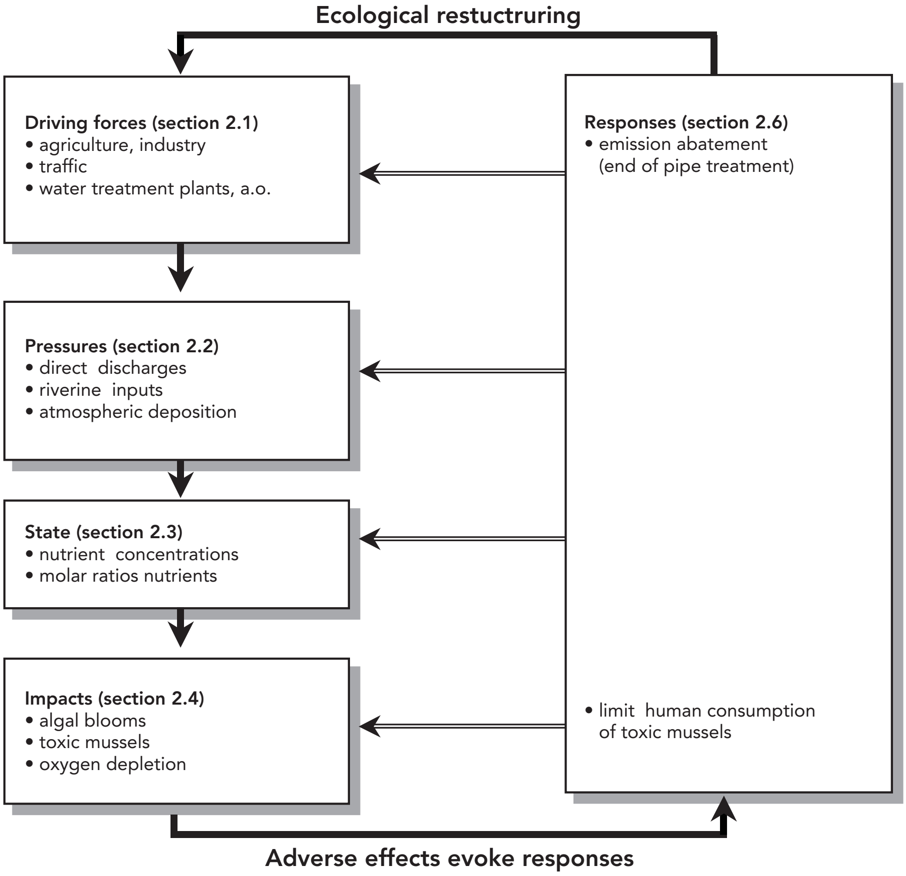
The Water Framework Directive (EU, 2000) stresses that the ultimate aim is to achieve concentrations in the marine environment near background values for naturally occurring substances like nutrients. Indicators are needed to determine and communicate the effects of the directives and measures on the state of eutrophication of marine and coastal waters.

2.1.2. DPSIR assessment framework for eutrophication

Figure 2 gives an overview of the DPSIR assessment framework for eutrophication. Several economic activities in society form the driving forces and are responsible for substantial loads of nutrients in the environment. For instance use of fertiliser and manure in agriculture and effluents from communal and industrial wastewater treatment plants contribute to the emission of nutrients in a river basin. Traffic and energy production also contributes by emitting nitrous oxides to the atmosphere. At a more local level other economic activities can be of importance, for example tourism or fish farming in enclosed coastal areas (EEA, 1999b).

Along different routes and after a longer or shorter time these emissions lead to inputs into marine and coastal waters (pressures) and increased nutrient concentrations in the marine and coastal water environment (state) which may result in eutrophication effects (impacts), like algae blooms and oxygen depletion.

DPSIR assessment framework for eutrophication in coastal waters Figure 2



The impacts are the triggers that evoke responses. Eutrophication of coastal waters leads to increased growth of algae (blooms) and foaming in periods that the algae are dying and degrade. Toxic algae may grow, which makes the blue mussel unhealthy for human consumption. Eutrophication may also lead to oxygen depletion. Eutrophication affects marine diversity, fish and shellfish stocks as well as human health and the recreational use of marine coastal zones (EEA, 1999b).

Principally, responses are possible at all levels in the DPSIR framework. But at the pressure and state level, measures are technically and economically hardly feasible (and also not wanted). Since policies aim at avoiding pollution and not at cleaning up and mitigating effects, measures should preferably be taken at the level of the driving forces. Emission abatement (treatment of wastewater and flue gasses) will result in decreasing emissions to the environment.

Sustainable development aims at ecological restructuring processing to minimise input of raw materials and energy as well as emissions to the environment. As Johnston et al. (2000) state: ‘Substances from the earth’s crust and substances produced by society must not systematically increase in the ecosphere’. For instance, in agriculture and cattle breeding ecological restructuring may be realised by increasing the recycling of organic materials to minimise fertiliser use, and in the transport sector cars low in emission of nitrous oxides may be developed. These responses are wanted because in the long run the economy is dematerialised and may grow without affecting ecology.

Eutrophication levels vary of natural causes from area to area. The productivity in the open eastern Mediterranean Sea is very low (oligotrophic), and in the open Baltic Sea relatively low (oligotrophic-mesotrophic), compared to the North Sea (mesotrophic) or upwelling areas, where nutrient-rich deep water comes to the surface.

2.2. Pressures

2.2.1. Sources of emission of nutrients

Several human activities contribute to the emission of nutrients to the environment (soil, water and air). Fertiliser industries discharge pollutants with their wastewater.

Effluents from sewage treatment plants also contain nutrients. Use of fertiliser and manure on agriculture land leads to run-off of nutrients to surface water and to emissions to the atmosphere. Traffic and energy production also emit nitrogen compounds to the atmosphere. The emissions to the atmosphere (also from waste burning) result in deposition of nitrogen compounds on river-basin areas and on marine and coastal waters.

Gathering information on the land-based sources contributing to nutrient emissions does not belong to the tasks in the project aimed at writing this report. Therefore the emissions of nutrients of land-based sources are not elaborated.

It must be kept in mind, however, that emissions from different (land-based) sources can be related to EU policies and targets set on abatement of nutrient emissions, meaning that emission indicators might be very helpful in clarifying the relation between human activities (driving forces) and the state of the marine and coastal environment.

2.2.2. Input routes of nutrients to coastal and marine waters

Nutrients are brought to the marine and coastal water environment by several sources and along different input routes:

- direct waterborne discharges in marine and coastal waters (direct inputs to water). These concern coastal discharges of industries and municipalities, waste discharges at sea, and emissions at sea, for example from passenger shipping. The nutrient-rich wastewaters mix with the coastal water leading to nutrient concentrations decreasing with the distance from the discharge location;
- loads of nutrients entering coastal waters with river inflow (riverine inputs). Both human activities as well as natural sources in the whole river basin may contribute to this input. The freshwater volumes from rivers spread out when flowing into coastal waters and form river plumes (see for example Szekielda and McGinnis, 1990). Near the coast freshwater is found. Salinity increases with the distance to the coast due to mixing with seawater by tidal and local currents. As a result of this mixing gradients in nutrient concentrations are also observed. When the river plume due to tidal currents moves along the coast, a 'coastal river' will be formed with gradients in salinity and nutrient concentrations in a direction perpendicular to the coast;
- wet and dry precipitation on marine and coastal waters (atmospheric inputs). For atmospheric inputs of nutrients, only nitrogen (e.g. nitrous oxide) is of importance. Again both human activities and natural sources may contribute to this input. Dry precipitation of nitrogen increases the concentrations in marine and coastal waters. In case of wet precipitation it depends on the concentration of nitrogen in rain water compared to the concentration in marine and coastal waters.

2.2.3. Availability and reliability of data on direct and riverine inputs of nutrients

OSPAR collects data on inputs of nutrients (nitrate-N, total N, phosphate-P and total P) into the North-East Atlantic including the North Sea on a yearly and systematic basis (OSPAR/ASMO, 1998). Helcom systematically collects data on inputs of nutrients (total N and total P) into the Baltic Sea on a five-yearly basis (Helcom, 1998). From 2000 onwards data will be collected annually. Medpol has data on loads of dissolved nutrients (nitrate-N, ammonium-N and phosphate-P) into the Mediterranean from major rivers. Medpol also collected information through different projects on the sewage discharges from the main Mediterranean coastal cities. The difficulty in collecting data has resulted in fragmented information with findings of little practical use. The objectives of Medpol phase III include an assessment of all sources of pollution reaching the Mediterranean Sea (UNEP/EEA, 1999). Nygaard et al. (2001) give an overview of the data that are requested and the data that are actually available.

The input data of OSPAR are not complete. Time series of Portugal and Spain for the Bay of Biscay and Iberian coasts are incomplete and highly irregular. Some data on inputs of 1997 and 1998 are missing and the input data of 1997 and 1998 are not checked as yet. The Helcom input data of nutrients are partly estimated and incomplete (Helcom, 1998). UNEP/EEA (1999) states for the Mediterranean that quality assurance and control procedures should be further developed and implemented to ensure data quality and reliability.

The reliability of annual river load estimates is largely determined by the sampling strategy. The reliability of the estimates can differ significantly for different substances and rivers (De Vries and Klavers, 1994). Another problem is the difference between gross riverine inputs and net inputs when the river crosses an estuary before entering coastal waters. In the estuary the load of nutrients may change due to denitrification and settling of nutrients with suspended solids (ICES, 2000). The gross inputs are calculated, but for the state of coastal waters and the impacts the net inputs are relevant.

The loads of substances transported yearly with rivers into coastal waters are related to yearly river flows. In years with high river flows, input loads are higher than in years with low river flows. The relation between flow and input loads, however, is not linear. The nature of the relation depends on the substance (solubility in water and adsorption to suspended solids, contribution of natural sources to the loads), river characteristics (rain-fed or glacier river, flow velocities, flow distribution within a year, frequency and extent of flushing events and suspended solids concentration), and river-basin characteristics (population, landscape and slopes and land use within the river basin). At present an unambiguous relation between river flow and nutrient loads cannot be given. Probably, the relation will be different for each river.

When the concentration of a substance is below the detection limit, two load estimates (upper and lower values) are calculated by OSPAR and Helcom. The lower estimate is calculated such that any result below the detection limit is considered equal to zero, while for the upper estimate, the value of the detection limit is used.

2.2.4. Availability and reliability of data on atmospheric inputs of nutrients

OSPAR collects data on atmospheric inputs of nitrogen. Measured data from 1987 onwards are available from the Comprehensive Atmospheric Monitoring Programme (CAMP). The coverage of measuring stations as well as time series of monitoring data are best around the North Sea. To estimate atmospheric inputs of N modelling work is carried out comprising transport and deposition of acidifying and eutrophying substances (OSPAR/ASMO, 1998). Helcom (1997) collects data on atmospheric inputs on a yearly base (see also EMEP, 2001). UNEP/EEA (1999) pays no attention to atmospheric inputs of nutrients making clear that data on atmospheric inputs are not collected for the Mediterranean.

There are large uncertainties in the results of different studies on atmospheric inputs (Ducrotoy et al., 2000), making comparison with direct and riverine inputs difficult. But results from the same calculation method may be compared to each other, making trend analysis (in a relative sense) for atmospheric inputs only possible and reliable.

2.2.5. Results of trend analysis for direct and riverine input of nutrients

To analyse trends high and low levels of nutrient inputs reported by OSPAR are averaged and subsequently all averaged inputs are expressed as a percentage of the total averaged input of a nutrient in 1990. The results of trend analysis may be presented at different levels of aggregation. The highest level of aggregation provides an insight into the development of nutrient inputs into European coastal waters over time. At this level the nutrient loads are aggregated for the whole North-East Atlantic including the North Sea. Missing values for nutrient inputs from countries in the last years of the period bias the trend analysis. Therefore in case of missing values the input values of the preceding year are used.

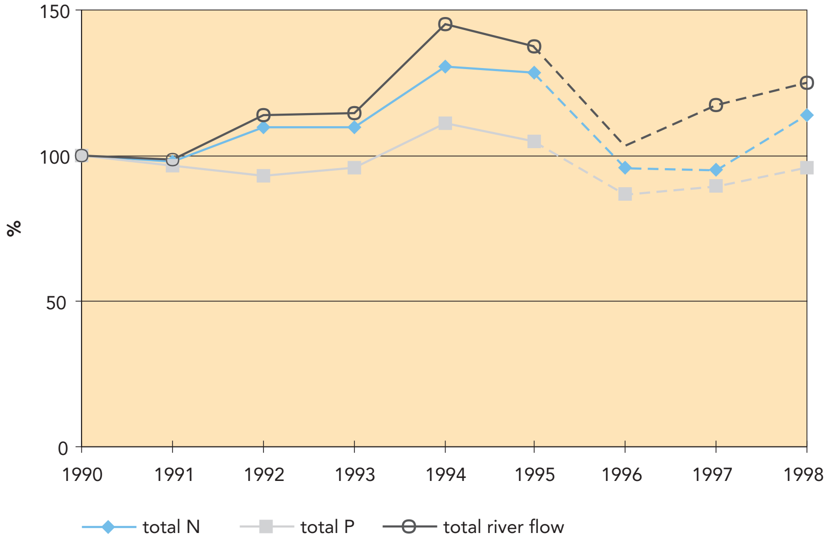
Fully aggregated

Results for the highest level of aggregation are presented in Figure 3. In this figure the development of loads of total N and total P over time as well as the total river flow are shown. Clear increasing or decreasing trends in the inputs are not found.

The course of the curves for direct and riverine inputs of nutrients is similar to that for the total river flow, indicating that natural variations may have a significant effect on the inputs of nutrients. In Figure 4 the yearly total inputs of nutrients are divided by the yearly total river flow, yielding average river concentrations. The concentrations calculated for 1990 have been given a value of 100 % and the concentrations for other years are derived from this percentage.

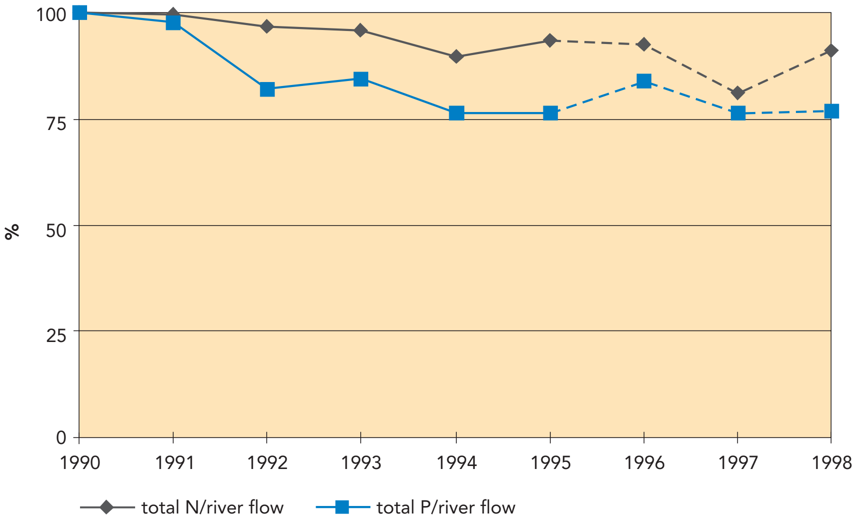
Figure 4 shows a clear decreasing trend for both total nitrogen and total phosphorus. Total phosphorus is decreasing more than total nitrogen.

Figure 3 **Change in loads (sum of direct and riverine inputs) of total N and total P into the North-East Atlantic including the North Sea in the period 1990–98 (input in 1990 is 100 %)**



Note: Time series on direct and riverine input of Spain is far from complete; therefore all input data of Spain are omitted. Some data from 1996, 1997 and 1998 are lacking and these data are provided by making estimates. For river flows average data are used of the years for which data are available. For inputs data of preceding years are used. Incomplete data and use of estimates are presented by dotted lines.

Figure 4 **Change in concentrations of nutrients in rivers flowing into the North-East Atlantic including the North Sea in the period 1990–98 (1990 values = 100 %)**



Note: See Figure 3.

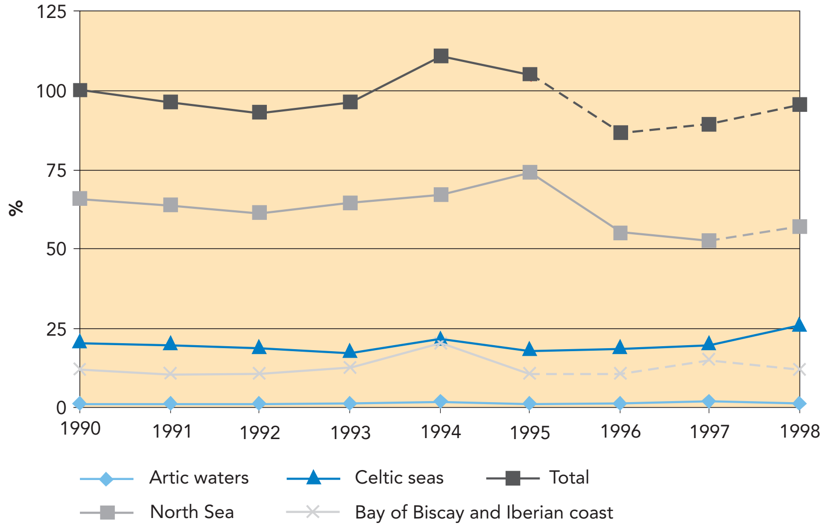
Regional seas

In Figure 5 the loads (sum of direct and riverine input) of total P into regional seas are presented. The North Sea accounts for the major part of the input of total P. This input into the North Sea clearly decreases between 1990 and 1998. The input of total P in the Celtic seas and in the Bay of Biscay and the Iberian coast does not show a clear trend. The input of P into the Arctic waters is small and hardly affects the total for all regional seas.

Considering total nitrogen, the change in inputs for the North Sea also compares favourably with the other regional seas. Between 1990 and 1998 the input of total N into the North Sea increases relatively less then the inputs into the other regional seas.

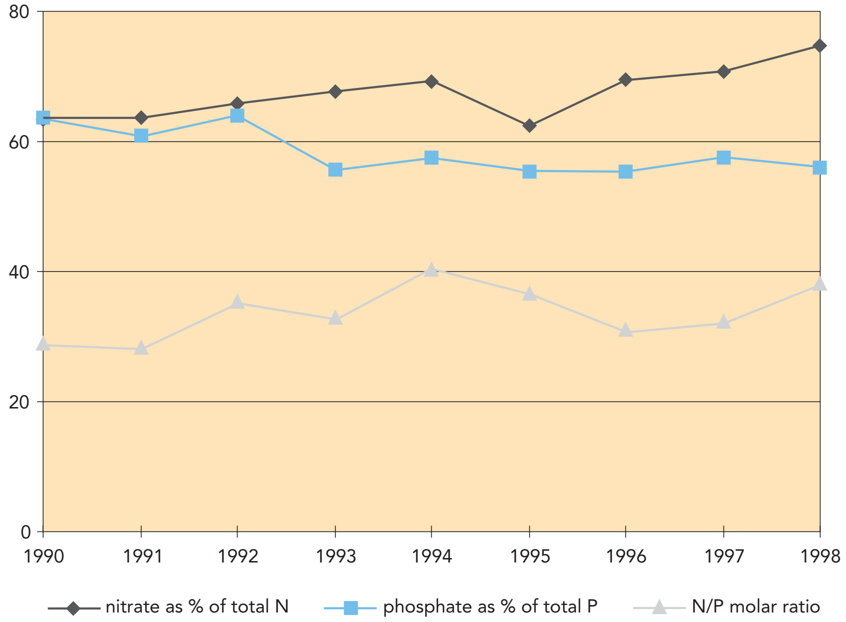
Comparison of the nutrient loads entering the Baltic Sea in 1990 and 1995 shows an increase in total nitrogen by 15 % and a decrease in total P by

Trend in sum of direct and riverine input of total P into the regional coastal waters of the North-East Atlantic including the North Sea in the period 1990–98 Figure 5



Note: See Figure 3.

Change in nutrient composition of inputs into the North Sea in the period 1990–98 Figure 6



18 %. However, because of considerable inter-annual variations in riverine run-off, these estimates are hardly comparable, also as the load data of 1995 was more complete and representative (Falandysz et al., 2000).

UNEP/EEA (1999) states that the nutrient levels found in Mediterranean rivers are about four times lower than those in western European rivers. Nitrate levels are increasing. Phosphate concentrations may increase dramatically, as seen in Greece, or steadily as found in France, or decrease when phosphate restriction measures are imposed as in Italy.

Country contributions

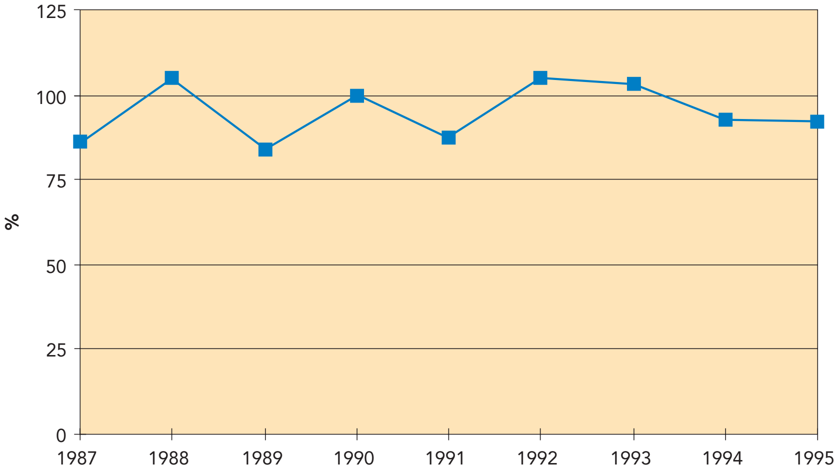
At a still lower level of aggregation country contributions to nitrogen and phosphorus input may be presented, showing changes over time (see Figure 18 for zinc as an example).

Composition of direct and riverine inputs of nutrients

For eutrophication in coastal waters not only the inputs of total nitrogen and total phosphorus are important, but also the nutrient composition of the inputs. In Figure 6 the change in the nutrient composition of the input into the North Sea is

Figure 7 **Change in wet deposition of nitrogen into the North Sea (525 000 km²) in the period 1987–95 (wet deposition in 1990 = 100 %)**

Source: OSPAR/ASMO (1998).



presented for the period 1990–98. Since the input of total phosphorus into the North Sea decreases while the input of total nitrogen slightly increases, the average N/P molar ratio in the input increases over time. The nitrate share in the input of total nitrogen increases in the period 1990–98, while the phosphate share in the input of total phosphorus decreases.

2.2.6. Results of trend analysis for atmospheric inputs of nutrients

OSPAR/ASMO (1998) used wet deposition measurements to calculate wet deposition of nitrogen into the North Sea (surface area 525 000 km²) in the period 1987–95. Extrapolation and model calculations by method 2a resulted in a deposition ranging from 298 to 371 kton a year (yearly average 337 kton a year). A temporal change is not observed (see Figure 7).

The yearly average of direct and riverine input of total nitrogen in the North Sea in the period 1990–98 amounts to 1 137 kton a year. Atmospheric inputs account for 23 % of the sum of both inputs, a significant share which may not be neglected on the scale of the North Sea as a whole, even not when we know that input values calculated for atmospheric deposition are less reliable than values for direct and riverine inputs. However, the share of atmospheric inputs is much smaller in the coastal waters where riverine inputs have most effects (confined to about 20 km from the coast). In these coastal waters atmospheric inputs may be neglected. However, Pryor and Barthelmie (2000) state that although fluvial pathways typically dominate the annually averaged nitrogen flux to coastal waters, atmosphere-surface exchange represents a significant component of the total flux and may be particularly critical during the summertime when both the riverine input and ambient nutrient concentrations are often at a minimum.

The mean deposition of total nitrogen in the entire Baltic Sea was estimated at 324 000 tonnes annually in 1986–90 (Falandysz et al., 2000), accounting for about 25 % of the total input. Recent estimates show that there was a decrease in the nitrogen deposition into the Baltic proper by about 20–30 % and into its drainage area by about 10–25 % from the mid-1980s to 1995 (Falandysz et al., 2000).

Bashkin et al. (1997) calculated the deposition of airborne anthropogenic nitrogen compounds on the Mediterranean Sea using models. The computed value amounted to 1084 kton N in 1992 and includes contributions from African and other non-EEA countries. Direct and riverine input was estimated at 800–1 200 kton N. According to these figures atmospheric deposition of nitrogen accounts for about half of the total nitrogen input in the Mediterranean Sea.

Atmospheric deposition of phosphorus is small when compared to direct and riverine inputs (even on the scale of the whole North Sea) and may be neglected.

2.2.7. Usefulness of inputs of nutrients as pressure indicator

The input indicator provides an insight into the development of inputs of nutrients over time. At high levels of aggregation, results of trend analysis provide an insight into the development of nutrient inputs into coastal waters on the European scale. Presenting results at lower levels of aggregation may show developments in inputs for regional seas or country contributions to inputs.

For riverine inputs of nutrients, river flow, nutrient concentrations and input loads are related. What the best parameter is for riverine input to describe the pressures is a question still to be answered. The direct and riverine inputs of nutrients are more sensitive to yearly natural variations in river flow than the average concentrations of nutrients in rivers. A lower sensitivity to natural variations makes a variable more valuable as an indicator, since for trend analysis shorter time series are needed. That means that the average river concentrations might be preferred as pressure indicators.

2.2.8. Recommendations for improving the quality of pressure indicators on nutrients

Guidelines for monitoring exist (OSPAR/ RID, 1998; Helcom, 1998) but improvement remains possible and harmonisation on a European scale is recommended. UNEP/ EEA (1999) reports that development of an effective, common Mediterranean monitoring system of measurements of contaminants and their effects is still missing, although monitoring in the Mediterranean has been in place for a long time. UNEP/ EEA (1999) also states that in the Mediterranean quality assurance and control procedures should be further developed and implemented to ensure data quality and reliability. ICES (2000) indicates that efforts to standardise data collection and dissemination need to be encouraged and enhanced and that there is a need for improved, integrated monitoring through coordination and harmonisation of existing national and international monitoring activities, as well as through implementation of new methods and technology.

For rivers flowing into estuaries the difference between the gross inputs of nutrients as calculated and the net inputs actually entering coastal waters may be significant. Where rivers flow into estuaries net inputs should preferably be monitored instead of gross inputs.

Simulation models are often used to determine atmospheric inputs, which make the atmospheric input data comparable to each other but often not directly to direct and riverine inputs. Further improvement of methods to determine atmospheric inputs and to integrate these inputs geographically with direct and riverine inputs is recommended (see also Pryor and Barthelmie (2000) who describe the major uncertainties in the measurement and modelling of atmospheric deposition of nitrogen).

When considering regional seas as a whole and not only coastal waters, developments of atmospheric inputs of nitrogen over time are also important. Therefore, to improve the usefulness of the input indicator on nutrients, atmospheric inputs of nitrogen may be included in addition to the waterborne loads.

2.3. State

2.3.1. Nutrient concentrations

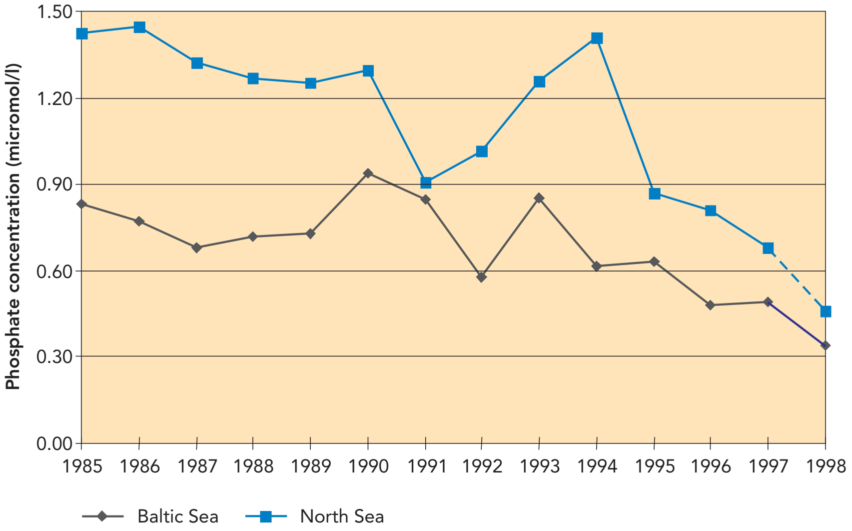
Inputs of nutrients into coastal waters spread through the coastal water environment. In case of riverine inputs, mostly the major part of inputs into coastal waters, a river plume is formed. Due to mixing with coastal water, the salinity of the water in the plume increases with the distance to the coast, while nutrient concentrations decrease. At the rim of the plume the salinity and the nutrient concentration become equal to those in the surrounding coastal water. Large riverine flows result in large river plumes (or coastal rivers) and a large area in which the riverine nutrient inputs may have effect.

Nutrient concentrations are monitored in European coastal waters, both in winter and year-round (Ærtebjerg et al., 2001). In the winter period the nutrient concentrations are higher and less variable than in the summer period. In the summer period nutrients are used for primary production, resulting in a decrease of the nutrient concentrations. For this reason the winter period is preferred to use for indicator purposes. From the data available on nutrient concentrations, the molar N/P ratio and other information on the composition may be determined easily.

2.3.2. Availability and reliability of data on nutrient concentrations

The data on nutrient concentrations provided cover the period 1985 to 1997/98 and contains median, maximum and minimum values on a winter, summer or year-round basis. The ICES data are medians per 10 x 10 km squares within 20 km from the coast. The data coverage was highest for winter surface concentrations of inorganic nitrogen (sum of NO_3, NO_2 and NH_4) and phosphate. The geographical data coverage is by far the best for the Helcom and OSPAR areas. Data from the northern Atlantic, the Bay of Biscay and the Iberian coast are sparse,

Figure 8 **Change in average winter phosphate concentration in the Baltic Sea and the North Sea in the period 1985–98**



Note: The average values for 1998 are based on only a few data (for the Baltic Sea only one), meaning that the 1998 values are less reliable

and the Mediterranean Sea is generally poorly covered (Ærtebjerg et al., 2001). Nygaard et al. (2001) provide an overview of the data requested and the data reported to the EEA.

Ærtebjerg et al. (2001) stress that a minimum time series of five years is required before a trend can be statistically detected at a 5 % significance level. Time series with this minimum length are only available for the Baltic Sea and the North Sea. In the North Sea, nutrient concentrations have large inter-annual variations. Winter nutrient concentrations in the North Sea are dominated by the outflow from the large European rivers (Seine, Meuse, Rhine, Ems, Weser and Elbe rivers). Outflow from these rivers is governed by precipitation over western and central Europe in wintertime, which can vary by a factor of two to three (Ærtebjerg et al., 2001).

2.3.3. Results of trend analysis for nutrient concentrations

The trend analysis is applied using the ICES data on winter concentrations of nitrate and phosphate in the coastal waters (20 km width) of the Baltic Sea and the North Sea.

The concentration level of nitrate and phosphate in winter differs between the Baltic Sea and the North Sea as does salinity. For this reason both regional seas are dealt with separately in trend analysis. For each sea and for each year the median concentrations observed at the ICES squares are averaged giving yearly average values independent of the locations. These average values are presented in Figure 8 for phosphate and in Figure 10 for nitrate.

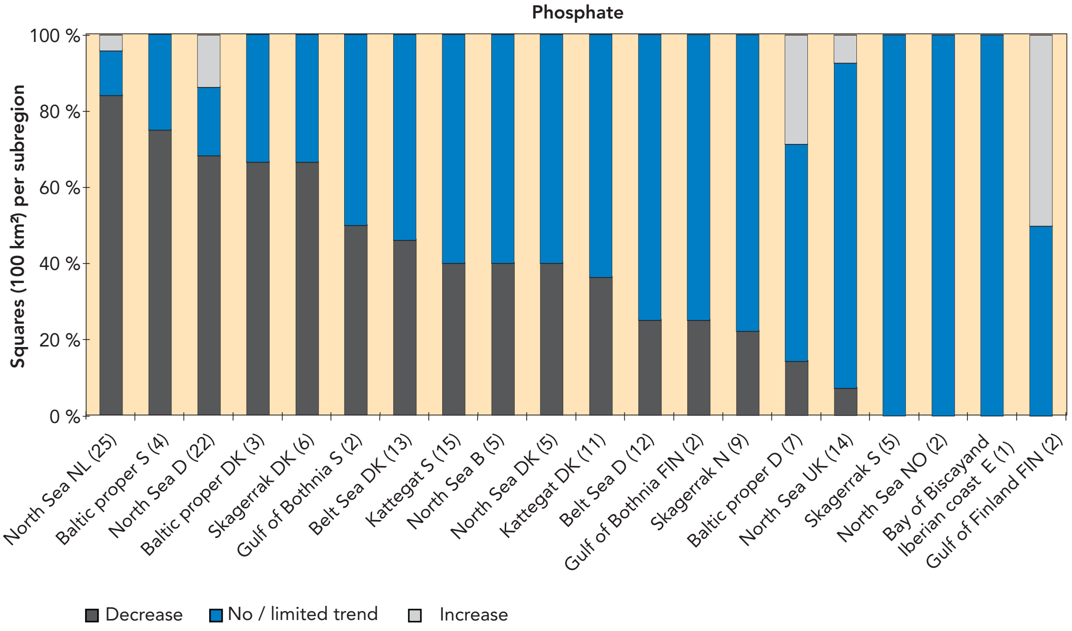
Phosphate

From Figure 8 it becomes clear that the natural variations in the yearly average concentration of phosphate in the winter period are large. The temporary increase in phosphate concentrations in the North Sea in the period 1991–94 coincides with increasing total river flows and subsequent increased inputs of phosphorus (see Figure 3). Altogether, a clear decrease in phosphate concentrations in both seas may be observed between 1985 and 1997. For the North Sea where phosphate concentrations are higher than in the Baltic Sea the rate of decrease in the period 1985–98 is larger than for the Baltic Sea.

Figure 9 presents results of trend analysis for phosphate concentrations in winter periods at a lower level of aggregation. For each of 20 coastal waters changes in phosphate concentrations are given in the period 1985–97. For each square (10 x 10 km) the change in phosphate concentrations in the period 1985–97 is determined. After that the squares are divided into three classes:

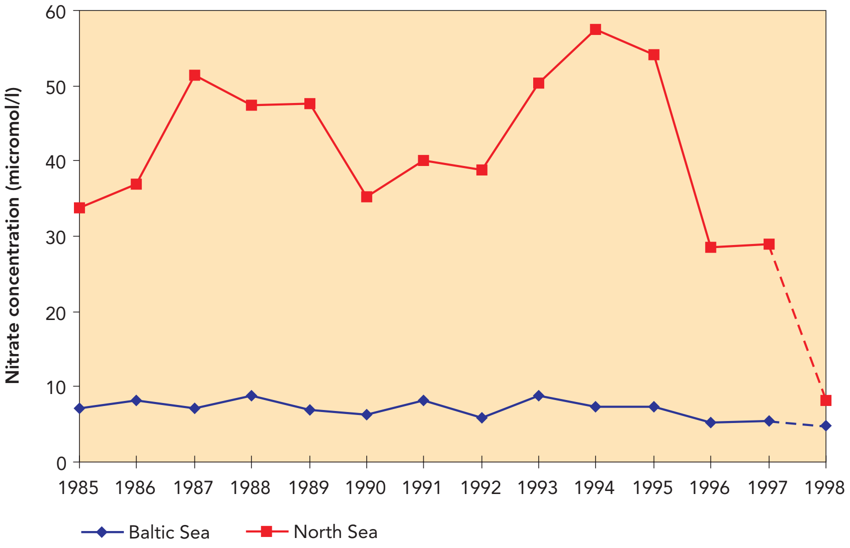
- more than 10 % decrease;
- more than 10 % increase;
- in between both.

Change in mean winter phosphate concentrations in coastal waters in the period 1985–97 Figure 9



Source: Van Buuren et al., 2002.

Change in average winter nitrate concentration in the Baltic Sea and the North Sea in the period 1985–98 Figure 10



Note: See Figure 8.

From Figure 8 it becomes clear that about half of the coastal waters show little or no change in phosphate concentrations. In 45 % of the coastal waters the phosphate concentrations improve significantly, while they worsen in about 5 % of the coastal waters. Thus it may be stated that on average the phosphate concentrations decrease, which is in agreement with the decrease observed in Figure 8.

Nitrate

In the period 1985–98, the average concentration of nitrate in the winter period also decreases in the North Sea and Baltic Sea, but less than for phosphate (see Figure 10). Van Buuren et al. (2002) present results of trend analysis for nitrate concentrations in the coastal waters in the winter periods in 1985–97 similar to Figure 9 for phosphate. For nitrate less improvement is observed, which is in line with the results presented in Figure 10 compared to those of Figure 8.

N/P molar ratio

Based on Figures 8 and 10 it may be concluded that the N/P molar ratio increased in the coastal waters of the North Sea and the Baltic Sea in the period 1985–98. In the North Sea this ratio *increased to a winter value of about 35 in 1996 and seems to decrease since then in winter periods (Kabuta and Duijts, 2000).*

2.3.4. Usefulness of nutrient concentrations as state indicators

The winter nutrient concentrations seem useful as eutrophication state indicators since trend analysis gave reasonable results for these variables and monitoring of these physical-chemical variables is relatively simple and already rather widespread.

The natural concentrations of nutrients differ between the European coastal waters depending on the local conditions (flushing time, salinity, tide and water dynamics, etc.). Therefore trend analysis may preferably be applied at the geographical level of the regional seas (the Baltic Sea, the North Sea, the Adriatic Sea, etc.).

2.3.5. Recommendations for improving nutrient concentrations as state indicators

Presently, data available on nutrient concentrations covers mainly the Helcom and OSPAR areas. The northern Atlantic, the Bay of Biscay and the Iberian coast, and the Mediterranean Sea are generally poorly covered. Completing the data set by improving the geographical coverage is recommended.

It is recommended to determine different classes of water quality describing the seriousness of the eutrophication situation depending on the natural circumstances. Comparison of different coastal waters is made easier in this way. Moreover, it is in agreement with the Water Framework Directive (EU, 2000), which makes a distinction between three quality classes of marine and coastal waters.

2.4. Impacts

2.4.1. Eutrophication in European coastal waters

Primary production enhances due to excess supply of nutrients from human activities. The increased phytoplankton production makes the water less transparent and leads to changes in the microalgal communities, i.e. the disappearance or decrease of long-lived species, and increases in the amount of opportunistic filamentous algae. Today, drifting filamentous algal mats are often observed. These algal mats are a hindrance to fisheries and a nuisance to recreational activities in many areas, but also affect other parts of the coastal ecosystem such as the seagrass meadows and shallow benthic animal communities. There is a large natural annual phytoplankton variability, but intensity of phytoplankton blooms, including those of toxic algae, may be a general indicator of primary production increase. The increased production and sedimentation of plant biomass may also lead to increased oxygen consumption in the deep soft-bottom areas affecting benthic communities.

Eutrophication effects are observed in many European coastal waters. For instance, in Norwegian coastal waters an increase in blooms of harmful algae is observed, primarily along the south coast of Norway, but also in local areas further north. Local or regional incidences of toxic mussels have been more frequent, and in later years some of these blooms have caused considerable mortality to salmon and trout in fish-farming cages as well (Skei at al., 2000).

In the Baltic Sea, eutrophication has favoured the blue mussel, as the increasing phytoplankton production provides food and limits the distribution of the bladder wrack. Thus the deeper rocky bottoms on which the bladder wrack was growing earlier are now covered by the blue mussel instead (Falandysz et al., 2000).

In the North Sea exceptional algal blooms have occurred and, while some have been a nuisance, others have been toxic or have contributed to increased organic enrichment. A general shift from long-lived macrophytes to short-lived nuisance algal species has occurred at various sites, requiring that about 50 000 tonnes of algae are removed each year from North Sea beaches (Ducrotoy et al., 2000). Each year, up to 15 % of the Wadden Sea is covered by algae, though with an apparent positive effect on zoobenthos productivity. While these algal mats are generally unsightly, they can also result in anaerobic conditions, the increase of opportunistic infaunal populations and the prevention of feeding by wading birds and fishes. Oxygen depletion may also adversely affect sediment deposition areas like the German Bight, with consequent benthos and fish kills (Ducrotoy et al., 2000).

Almost all coastal countries in the Mediterranean are affected by eutrophication, although at different levels (UNEP/EEA, 1999). The most important cases of eutrophication are found along the northern and western coasts of the Adriatic Sea, where dystrophic phenomena (heavily stained with high content of organic matter) are observed, causing very severe damage both to fishing and tourism (Cognetti et al., 2000).

2.4.2. Eutrophication effect variables

Although the effects of eutrophication are well known, the mechanisms governing its effects are poorly understood. In particular, effects on microbial processes are key to many aspects of the functioning of the ecosystem, and are commonly inadequately addressed (Meyer-Reil and Köster, 2000).

Ærtebjerg et al. (2001) present an overview of the eutrophication variables monitored in European coastal waters. Apart from nutrient inputs and nutrient concentrations, this involves:

- physical-chemical variables:
  - oxygen concentration in bottom layer. When oxygen falls below 125 µmol/l, benthic fauna is affected;

- biological variables:
  - phytoplankton (algal blooms, algae composition and chlorophyll-a);
  - benthic vegetation (seagrasses and seaweeds);
  - benthic fauna (microphytobenthos biomass and macro-zoobenthos biomass).

The oxygen concentration near the bottom is related to microbial activity, which in turn is stimulated by the supply of organic material and nutrients. Phytoplankton production is also related to eutrophication, and chlorophyll-a may be considered a relevant eutrophication effect variable. Though benthic vegetation and benthic fauna are sensitive to eutrophication, other factors may be more relevant for these biological variables. Therefore these biological variables are discussed in Chapter 5 on integrated biological indicators.

The oxygen content in bottom waters is determined by two processes (see Annex 1):

- the microbial consumption of oxygen due to degradation of organic materials in bottom water and sediment. The consumption rate depends on the amount and quality of organic material settling to the bottom and on the temperature;
- the supply of oxygen from vertical mixing and horizontal transport processes. The supply rate depends on hydrodynamic processes forced by wind, buoyancy and tides.

Oxygen deficiency will occur if the consumption rate exceeds the supply rate for a sufficiently long period of time and may happen in marine waters with periodic or permanent strong stratification.

Chlorophyll-a is the green plant pigment used for photosynthesis, which is present in all autotrophic phytoplankton organisms. Chlorophyll-a is an adopted measurement technique for monitoring the amount of algae.

2.4.3. Availability and reliability of data on bottom oxygen concentrations

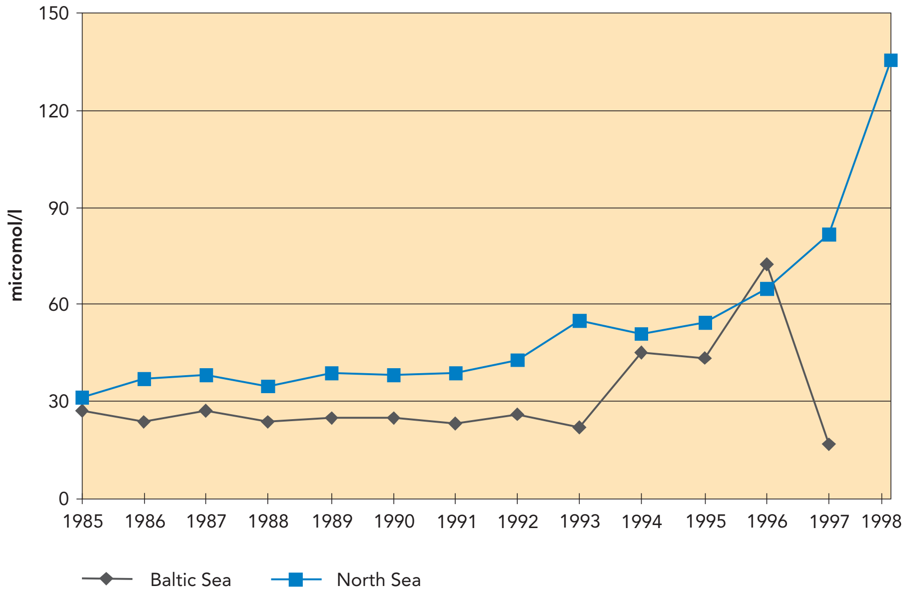
The Baltic proper, the Kattegat and the German Bight are the areas most susceptible to oxygen deficit due to their stratified waters and high sedimentation rates. In other coastal waters, tides and wind stress will ensure well-oxygenated conditions at the seabed (Ærtebjerg et al., 2001).

Data of mean September/October oxygen concentrations are available for the North Sea and the Baltic Sea. The data predominantly cover the period 1988–96 (see Nygaard et al., 2001). Methodology and frequency of data collection follow OSPAR and Helcom requirements subject to the ICES reporting format.

Data from the northern Atlantic, the Bay of Biscay and the Iberian coast are sparse. For the Mediterranean it is reported that anoxic conditions occur near the bottom at many locations (UNEP/EEA, 1999), but data and time series are not available.

Bottom oxygen concentration can vary markedly from day to day. Routine measurements will fail to detect episodes of sudden oxygen deficit. The frequency of sampling (maximum once every two weeks) does not always allow for monitoring of the occurrence of anoxic events in detail. This means that the period of oxygen deficit cannot be determined and neither can the bottom area in which it occurs. The available data reveal the problem areas and have a signalling function (Van Buuren et al., 2002).

Figure 11 **Trend in median autumn bottom oxygen concentrations in the Baltic Sea and the North Sea in the period 1985–98**



2.4.4. Availability and reliability of data on biological variables

Helcom has a 15-year monitoring programme on plankton and benthos, but these data are at present not available in a central database. OSPAR has been collecting similar data for some years in their nutrient monitoring programme, but these data have not yet been entered in a database. Data on harmful algal blooms are collected regularly.

The following countries have supplied data to ICES on summer chlorophyll-a concentrations in squares (10 x 10 km) extending at least five years: Belgium, Denmark, Finland, Germany, the Netherlands, Norway, Sweden and the United Kingdom (Ærtebjerg at al., 2001). Climatic variations give rise to significant variations in these concentrations.

2.4.5. Results of trend analysis for bottom oxygen concentrations

Trend analysis is applied at three levels of aggregation covering:

- both the North Sea and the Baltic Sea;
- 15 coastal waters; and
- locations in Danish coastal waters.

Aggregated for the North Sea and the Baltic Sea

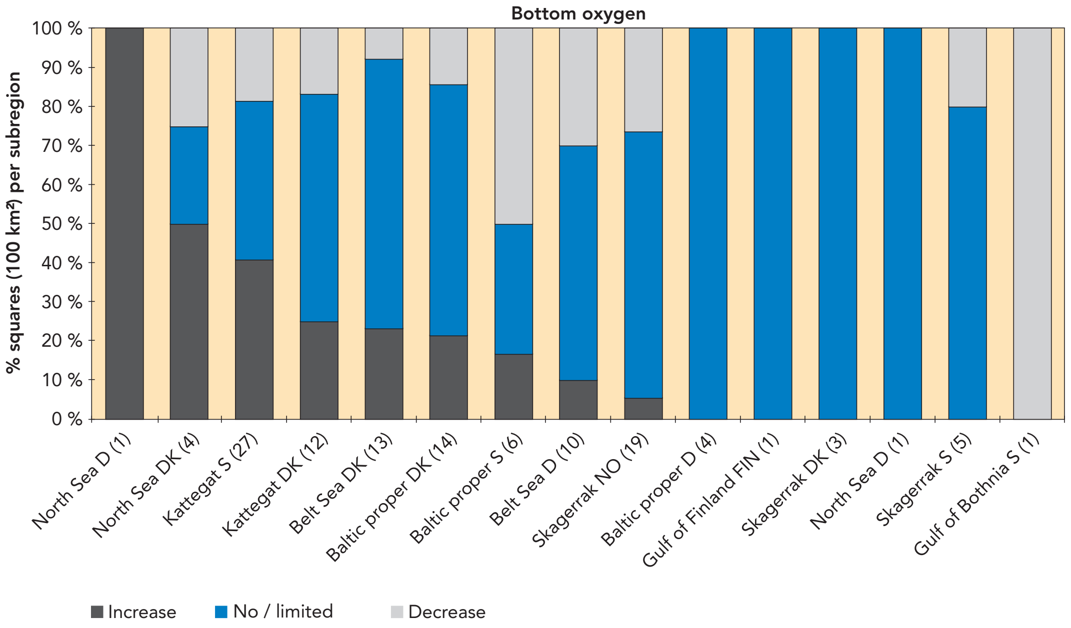
The data on mean bottom oxygen concentrations in September/October for stations in the North Sea and the Baltic Sea are used for trend analysis. In this period of the year, the risk of oxygen deficit is greatest due to decay of newly settled plant material. The level of bottom oxygen concentration in the autumn periods differs between the Baltic Sea and the North Sea. For this reason both regional seas are dealt with separately. In the first instance, for each of the seas and for each year, the mean concentrations observed at locations are averaged giving yearly average values independent of the locations. It turned out that the yearly average bottom oxygen concentrations calculated for the Baltic Sea are strongly affected by extreme high values observed at one location in the coastal waters of Sweden, for which the time series are not complete. To smooth out these irregularities, instead of mean values median values are calculated and presented in Figure 11.

Figure 11 shows that the autumn bottom oxygen concentrations in the North Sea do not change much until about 1992. But from 1993 onwards a clear improvement is observed. In particular the value for 1998 shows a major improvement, but it must be remarked that this value is based on less than half of the observations of the years before. The period from 1993 onwards is still too short to draw conclusions about a real positive trend occurring independent of natural variations.

In the Baltic Sea some improvement seems to occur from 1994 onwards, but again this is based on fewer observations for these years compared with the years before. It is too early to draw conclusions about the development.

Trends in bottom oxygen concentrations in coastal waters in the period 1988–96

Figure 12



Source: Van Buuren et al., 2002.

Coastal waters

In Figure 12 results of trend analysis are presented for the different coastal waters for the period 1988–96 (a shorter period than in Figure 11). Just like in Figure 9, three classes of change of bottom oxygen concentration are distinguished:

- 10 % or more increase;
- 10 % or more decrease; and
- values in between.

When viewing Figure 12 no clear improving trend in bottom oxygen can be found. More than 50 % of the squares show no or a limited trend, and decreases and increases are found in about equal numbers. The relative high values for 1997 and 1998 for the North Sea (see Figure 11) are not included in these results.

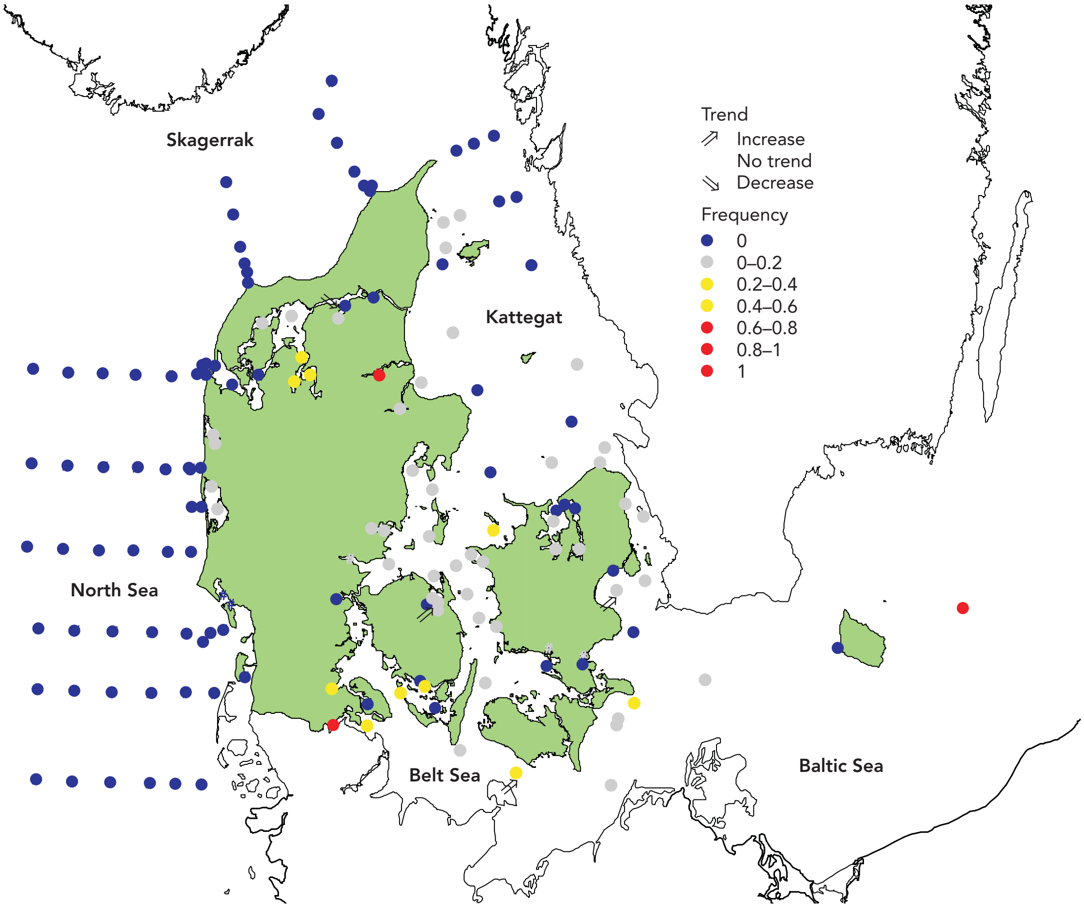
Case study on the indicator 'frequency of hypoxia' in Danish coastal waters

Annex 1 gives the details of the applied trend analysis on data on bottom oxygen concentrations of the Danish national marine monitoring programme. The number of hypoxic (oxygen concentrations below 2 ml/l) observations each year was considered to be binomial distributed with parameters n and p_i, where n is the total number of observations for year i and p_i is the probability of observing hypoxic conditions in year i. The trend analysis is conducted by testing if p_i is a function of year (logistic regression). A virtue of this method is that it weighs yearly frequency observations with the number of observations it has been based on. Thus, the frequency of hypoxia determined from a year with many observations of oxygen concentrations has more weight than the frequency of hypoxia determined from a year with few observations of oxygen concentrations.

The use of this indicator was exemplified with data from one station located in the southern Belt Sea between Germany and Denmark. This station was deliberately chosen because it showed an increasing frequency of hypoxia. An increasing trend in the frequency of hypoxia at 5 % significance level was found (P = 0.0183).

The frequency of hypoxia for the last half of the 1990s (1995 and onwards) and trends of this frequency (no specific period) is shown in Figure 13. Many stations can be characterised as stations without oxygen deficiency problems (frequency = 0). This applies especially to all the stations in the North Sea and Skagerrak, where tidal mixing and currents induce a constant replacement of bottom waters. In Kattegat, the Belt Sea and the Baltic Sea most stations are exposed to hypoxia, and these marine areas must be considered sensitive to oxygen deficiency. Some estuaries (typically estuaries with a sill) are exposed to almost permanent hypoxic conditions. The eastern-most station in the Baltic Sea is located in one of the deep Baltic Sea basins, where almost permanent hypoxia is observed from depths of 60–70 m down to the bottom at 86 m.

Figure 13 **Frequency of hypoxia for stations in Danish coastal waters and the change from 1995 onwards**



Logistic regression analyses of data from the period 1974–99 show that 4 out of 153 stations have a significant trend in the frequency of hypoxia (three stations show an increase and one station a decrease). Hence, a general trend in the frequency of hypoxia is not observed in this period. The period may be too short for trend detection and trends may be masked by natural variations.

2.4.6. Case study on the indicator 'chlorophyll-a' in Danish coastal waters

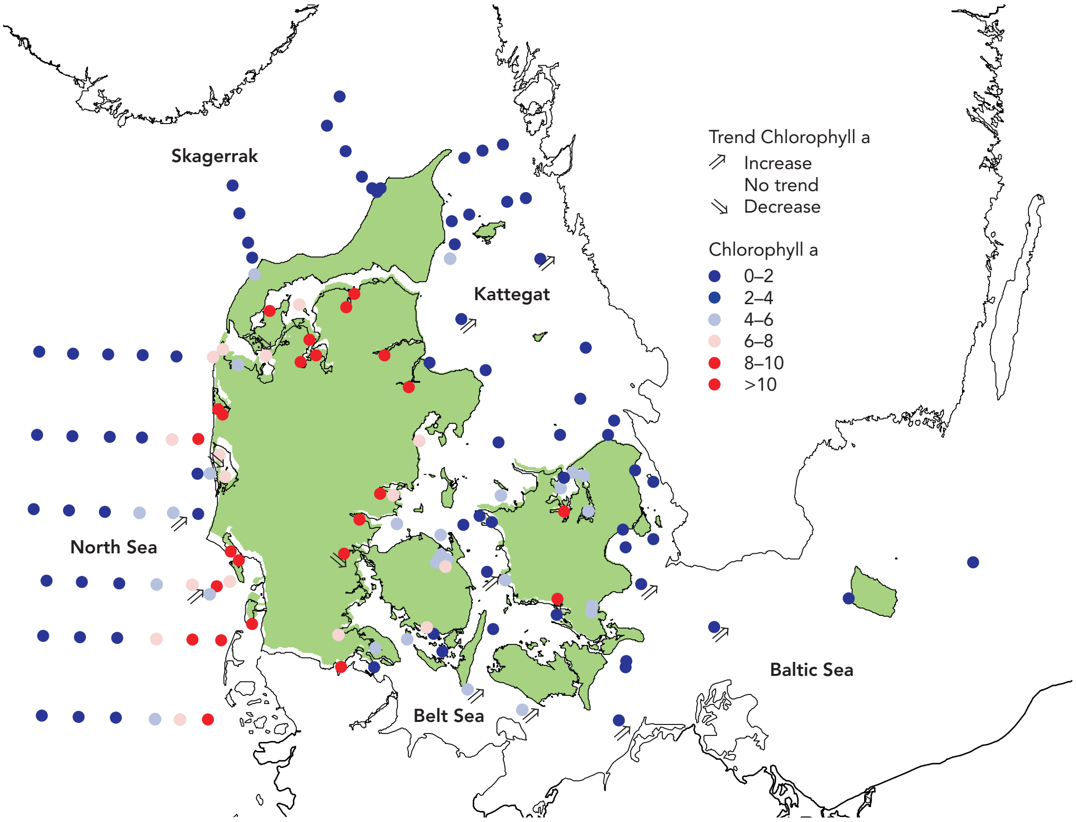
Phytoplankton responds quickly to changes in the limiting factors, i.e. nutrients and light. In periods with nutrient limitation and no light limitation the chlorophyll-a concentration is partly determined by the supply of nutrients, meaning that it may be used as an impact indicator for eutrophication.

The seasonal variation in phytoplankton biomass depends on the climatic conditions. Arctic waters have an intense summer bloom when light conditions are optimal. Temperate waters have a short intensive spring bloom, a summer period in which phytoplankton is mainly controlled by nutrient limitation and grazers, and an autumn bloom when wind mixing events bring nutrients to the surface waters. Subtropical waters have a weak seasonal variation; phytoplankton biomass does not change much over the year.

Hitherto, the chlorophyll-a indicator has been calculated on averaged depth values of 0–10 m, suitable for the North Sea and the Baltic Sea. In the Mediterranean Sea the productive zone stretches further down in the water column. Thus, the median summer chlorophyll-a (0–10 m) concentration from April to September is an inappropriate impact indicator of eutrophication when applied to all community marine waters.

It is proposed to use the average chlorophyll-a concentration in the period with nutrient limitation and no light limitation (see Annex 1). The depth and the time interval must be defined on a regional scale. For the North Sea and Baltic Sea 0–10 m of depth is considered appropriate and for the Mediterranean Sea a depth of, for example, 0–50 m seems more useful. The time period from April to September is an appropriate period for the North Sea and southern Baltic Sea, while this period is shorter in the Arctic waters and longer in subtropical waters.

Summer chlorophyll-a concentrations measured by the Danish national monitoring programme in 1999 Figure 14



Note: Arrows indicate significant trends determined with Kendall's τ (not for a specific period).

For trend analysis the average chlorophyll-a concentrations are used instead of the median, because chlorophyll-a concentration often has a skewed distribution in the nutrient limited period and median values neglect this skewness. Kendall's τ has been applied to the chlorophyll-a concentrations measured by the Danish national monitoring programme to detect trends. The results are presented in Figure 14.

From Figure 14 it may be observed that nutrient discharges from land-based sources in estuaries give rise to increased chlorophyll-a concentrations compared to the concentrations at coastal stations. The concentrations in the North Sea increase when approaching the German Bight.

The trend analysis shows that 14 out of 141 stations have a significant trend in summer chlorophyll-a concentration (10 stations show an increase and 4 stations show a decrease). Hence, there is no overall trend in the chlorophyll-a concentration. As with hypoxia, the period may be too short for trend detection and trends may be masked by natural variations.

Ærtebjerg et al. (2001) applied trend analysis on time series extending five years of summer chlorophyll-a concentrations of squares (10 x 10 km) in European coastal waters. For most of the locations a trend cannot be observed. Of the 105 locations analysed, only 4 locations show a decreasing trend, while 4 other locations show an increasing trend.

2.4.7. Usefulness of bottom oxygen, hypoxia and chlorophyll-a as impact indicator

Eutrophication is a complex phenomenon. The natural trophic state may differ locally in coastal European waters depending on the local conditions (flushing time, salinity, tide and water dynamics, etc.). Therefore trend analysis may preferably be applied on a regional and local geographical level (parts of the Baltic Sea, of the North Sea, and of the Adriatic Sea, etc.).

Natural variations in bottom oxygen concentrations and hypoxia, and in chlorophyll-a concentrations are large. That means that long time series are needed to determine trends, making these variables less valuable as an impact indicator. Bottom oxygen concentrations and the frequency of hypoxia may be used to identify marine and coastal waters susceptible to oxygen deficiency and thus sensitive to eutrophication. In these areas these variables can be used to monitor the development over time.

2.4.8. Recommendations for improving bottom oxygen, hypoxia and chlorophyll-a as impact indicator

Presently, data available on bottom oxygen concentrations and hypoxia cover coastal waters where oxygen deficiency is observed. Completing the data set by improving the geographical coverage to all European coastal waters where oxygen deficiency may be observed is recommended.

Apart from *in situ* measurements, remote sensing techniques can be used to determine chlorophyll-a concentrations in coastal waters. Simulation models can also be used to determine and predict eutrophication effects. It is recommended to combine these three information sources. By better integrating the data, more reliable results will be obtained, probably at lower costs.

It is recommended to determine different classes of eutrophication describing the seriousness of the eutrophication situation depending on the natural circumstances. Comparison of different coastal waters is made easier this way. Moreover, it is in agreement with the Water Framework Directive (EU, 2000), which makes a distinction between five quality classes of marine and coastal waters.

2.5. Links between pressure and state indicators

Some additional analysis is done on the pressure and state relationship by comparing riverine inputs of nutrients and concentrations of nutrients in the North Sea. Based on the results of the trend analysis presented in Figures 3, 4, 8 and 10 the relations between pressures (mainly riverine input of nutrients) and state (concentrations of nutrients in coastal waters) are twofold.

1. The decreasing trends in the average concentrations of nutrients in rivers entering coastal waters correspond well with the trends (smoothed for natural variations) in the median concentrations of nutrients in coastal waters.

2. The natural variability in riverine inputs of nutrients shows up clearly in the median concentrations of nutrients in coastal waters.

The explanation for these observations is slow mixing of river water (with relative high concentrations of nutrients) and marine water (with lower concentrations) leading to the formation of river plumes in coastal waters. The concentrations of nutrients in the river plume are mainly determined by the concentrations in the rivers. But how far the river plume reaches into sea depends on the river flow. Increasing river flows (and inputs of nutrients) result in larger river plume areas and thus more squares (of 10 x 10 km) where increased concentrations of nutrients can be found.

From these observations it may be concluded that concentrations of nutrients in rivers should preferably be used as pressure indicators. Then natural variations are smoothed out and trends may be detected rather easily.

For coastal waters where atmospheric deposition contributes significantly, as in the Baltic Sea, the atmospheric deposition should be taken into account before comparing deposition with nutrient concentrations.

2.6. Responses

Eutrophication is a major problem in many European coastal waters. The Baltic Sea States and the North Sea States have already decided on a 50 % reduction of the load with nitrogen and phosphorus compared to the late 1980s. The EU has also decided on measures to reduce eutrophication, for example. the Urban Wastewater Treatment Directive (91/271/EEC), the Nitrate Directive (91/676/EEC) and the Water Framework Directive (2000/60/EC). The proposed sixth action programme of the EC (2001) states that proper and full implementation of the Urban Wastewater Directive and the Nitrate Directive will be an important positive factor in reducing eutrophication.

Response indicators may be developed based on the degree of implementation of these directives and the resulting reduction in emissions of nutrients from land-based sources.

2.7. Options for further development of eutrophication indicators

The Water Framework Directive (EU, 2000) has the ultimate aim to contribute to achieving concentrations in the marine environment near background values for naturally occurring substances like nutrients. Proper and full implementation of the Urban Wastewater and of the Nitrate Directives will be an important positive factor in reducing nutrient emissions from land-based sources (EC, 2001). Response indicators are needed for the degree of implementation of these directives and the resulting reduction in emissions of nutrients from land-based sources.

To determine background values for nutrients' research is needed on natural nutrient loads and nutrient composition of rivers, as well as on natural variations in these loads related to variations in river flow. The same holds for nutrient concentrations and the nutrient composition in marine and coastal waters. A better understanding of the natural state depending on local conditions and natural variations will enable us to compare the actual state with the natural one and to classify coastal waters as the Water Framework Directive prescribes.

Further improvement of methods to determine anthropogenic and natural atmospheric inputs of nitrogen and to integrate these inputs geographically with direct and riverine inputs of nitrogen is recommended.

Eutrophication of coastal waters is a complex phenomenon. Though eutrophication is widespread in European marine waters, effects are observed mainly in coastal water areas vulnerable to eutrophication. For these areas monitoring of eutrophication variables makes sense.

The effectiveness of monitoring to determine eutrophication of marine waters might be improved. At present, predominantly *in situ* measurements are made. Remote sensing techniques can also be used to determine eutrophication effects like chlorophyll-a concentrations in coastal waters. Moreover, simulation models can be used to determine and predict eutrophication effects. It is recommended to combine and integrate these three information sources, making the data more reliable and probably obtaining results at lower costs.

3. Indicators for hazardous substances

3.1. Introduction

3.1.1. Policy questions

Hazardous substances are defined as substances or groups of substances that are toxic, persistent and liable to bio-accumulate and other substances or groups of substances that give rise to an equivalent level of concern (EU, 2000). Hazardous substances may be divided into heavy metals, persistent organic pollutants (POPs) and mineral oil (dealt with in Chapter 4).

Heavy metals are metals or metalloids which are stable and which have a density greater than 4.5 g/cm^3, namely lead, copper, nickel, cadmium, platinum, zinc, mercury and arsenic (EEA, 1999a).

POPs are chemical substances that persist in the environment, bio-accumulate through the food web, and pose a risk of causing adverse effects to human health and the environment (UNEP, 1998; UNECE, 1998).

Due to human activities, inputs of hazardous substances into European coastal waters increased especially in the 1970s and early 1980s. European policies aim at reducing the inputs and improving the state of the marine and coastal environment. Measures to reduce the input of hazardous substances are being taken as a result of various initiatives at all levels (global, European, national, and regional conventions and ministerial conferences): the UN global programme of action for the protection of the marine environment against land-based activities; EU directives: the Water Framework Directive (2000/60/EC), the Mercury Discharge Directive (82/176/EEC), the Cadmium Discharge Directive (83/513/EEC), the Mercury Directive (84/156/EEC) and the Dangerous Substance Discharges Directive (86/280/EEC); the Mediterranean action plan (MAP); the 1992 Helsinki Convention on the Protection of the Marine Environment of the Baltic Sea Area; and the 1998 OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic.

The target is a substantial reduction of the input of hazardous substances to coastal waters. In the OSPAR area the target is set at 50 % reduction of the emissions of several hazardous substances between 1985 and 1995 and zero emissions in 2020. The ultimate aim of the Water Framework Directive (2000/60/EC) is to achieve the elimination of priority hazardous substances and contribute to achieving concentrations in the marine environment near background values for naturally occurring substances. Indicators are needed to determine and communicate the effects of the directives and measures on the state of the marine and coastal environment. As targets have been formulated for emissions and discharges and not input loads into coastal waters, only an approximate comparison is possible between the measured input and the reduction target aimed at emissions.

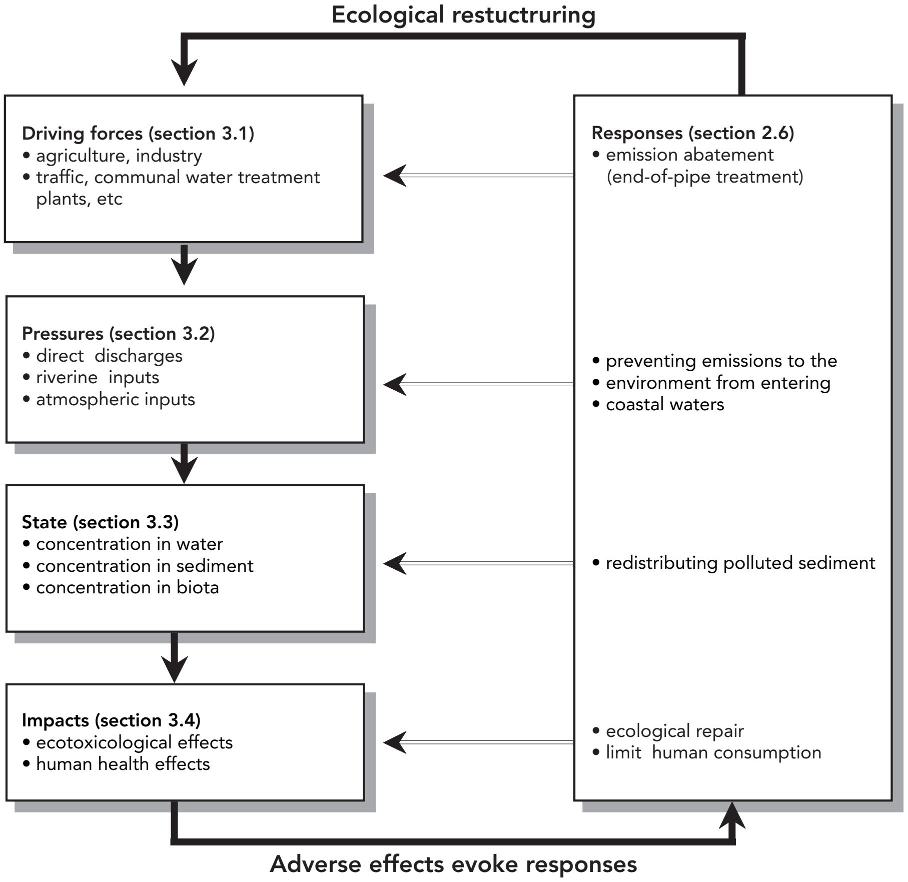
Emission sources and emission patterns differ between various hazardous substances. Thus, in determining the results of pollution abatement policies, each substance should also be considered separately. Due to the high (eco)toxicity of mercury and PCB$_7$, even at very low concentrations, these substances have received a great deal of attention and priority in international reduction measures.

3.1.2. DPSIR assessment framework for hazardous substances

The DPSIR assessment framework for hazardous substances is similar to that for eutrophication (see Figure 15). Several hazardous substances are produced by the (petro)chemical industry. Due to by-products formed and the use of chemicals in society, the environment is polluted with hazardous substances. The two main drivers behind the contamination of the environment are the satisfaction of demand for ever-newer consumer products, which may involve innovative chemicals, and the need to find uses and markets for the products and by-products of the (petro)chemical industry (EEA, 1998).

Coal combustion, waste burning and the metal industry are mainly responsible for the emission of hazardous substances to the air. Surface waters are polluted by run-off of agricultural land (fertiliser use containing heavy metals, use of pesticides for control of

DPSIR assessment framework for hazardous substances Figure 15



diseases and pests) and effluents from municipal and industrial wastewater treatment plants. Along different routes and after a longer or shorter time these emissions lead to inputs into marine and coastal waters (pressures) and concentrations in the marine and coastal water environment (state), which may result in eco-toxicological effects (impacts). Human health may be affected when contaminated seafood is consumed.

Adverse impacts are the triggers that may evoke responses. Principally, responses are possible at all levels in the DPSIR framework. But at the pressure and state level measures are technically and economically hardly feasible. Measures should preferably be taken at the level of the driving forces. Emission abatement (treatment of wastewater and flue gasses) will result in decreasing emissions to the environment.

3.2. Pressures

3.2.1. Sources of emissions of hazardous substances

Emissions of hazardous substances to the environment are related to the production and use of chemicals and metals in society. Specific uses such as mercury by dentists can be controlled by substance specific policies. These policies must be combined with generic policies, aimed at reducing emissions from, for example, wastewater treatment, waste burning and treatment of flue gasses.

Gathering information on the land-based sources contributing to the emissions of hazardous substances does not belong to the task in the project aimed at writing this report.

As targets have been formulated for the emissions and discharges of hazardous substances from land-based sources and not for riverine inputs into marine and coastal waters, emission indicators might be helpful in describing human impacts on the marine and coastal environment.

3.2.2. Input routes

Hazardous substances reach marine and coastal waters along three input routes:

- direct discharges in marine and coastal waters (direct inputs). These concern coastal discharges of industries and municipalities, waste discharges at sea, and

emissions at sea of, for example, shipping and oil production. Depending on the water volumes in which the hazardous substances are discharged and the concentrations of the substances in these volumes, the concentrations in marine and coastal waters may be increased locally near the discharge location;
- loads of hazardous substances entering coastal waters with river inflow (riverine inputs). Both human activities as well as natural sources in the whole river basin may contribute to this input. The water volumes from rivers flowing into coastal waters generally are large. Often, a river plume is formed flowing along the coast, fresh near the coast and increasing in salinity with the distance to the coast, describing the mixing of river water and seawater. As a result gradients in concentrations of hazardous substances are also observed;
- wet and dry deposition on marine and coastal waters (atmospheric inputs). Atmospheric transport of substances may take place over long distances. Again both human activities as well as natural sources may contribute to this input. Dry deposition increases the concentration of hazardous substances in marine and coastal waters. For wet deposition this depends on the concentration of these substances in rainwater compared to the concentration in marine and coastal water at the location where the deposition forms input.

The concentrations of hazardous substances in a river are related to the river flow (see paragraph 2.2.2). Depending on the characteristics of the substance and the fate in the environment the concentration may increase or decrease with river flow, or remain more or less equal.

3.2.3. Availability and reliability of data on direct and riverine inputs

OSPAR collects data on inputs of hazardous substances into the North-East Atlantic including the North Sea on a yearly and systematic basis (OSPAR/ASMO, 1998). Helcom systematically collects data on inputs of hazardous substances into the Baltic Sea on a five-yearly basis (Helcom, 1998). From 2000 onwards data will be collected annually. The OSPAR as well as the Helcom member countries, however, have so far not agreed on the same methods for the determination of the input loads. Medpol collects data on inputs of hazardous substances into the Mediterranean but states that the monitoring capabilities of some countries have to be improved (UNEP/EEA, 1999).

The input data of OSPAR is not complete. Some time series are very irregular, while others are questionably unchanging (Baan and Groenewegen, 2000). The Helcom input data of heavy metals into the Baltic Sea are partly estimated and incomplete (Helcom, 1998). Input data on lindane and PCB_7 are lacking. UNEP/EEA (1999) states that, for the Mediterranean, quality assurance and control procedures should be further developed and implemented to ensure data quality and reliability.

The reliability of annual river load estimates is largely determined by the sampling strategy. The reliability of the estimates can differ significantly for different substances and rivers (De Vries and Klavers, 1994). When rivers flow through estuaries, these estuaries act as filters, retaining part of the substances. As a result the net input into coastal waters is less than the gross inputs, the loads transported by the rivers (Zwolsman, 1994). Due to ongoing pollution abatement in the river basin oxygen concentrations in the river may improve. This has important consequences for the role played by the estuary as a filter for heavy metals. Restoration of oxygen conditions is expected to increase the outflow of dissolved heavy metals from the sediments in an estuary for a certain period of time (Zwolsman, 1999).

The loads of substances yearly transported with rivers into coastal waters are related to yearly river flows. In years with high river flows, input loads are higher than in years with low river flows. The relation between flow and input loads, however, is not linear. The nature of the relation depends on the substance (solubility in water and adsorption to suspended solids, contribution of natural sources to the loads), river characteristics (rain-fed or glacier river, flow velocities, flow distribution within a year, and frequency and extent of flushing events and suspended solids concentration), and river-basin characteristics (population, landscape and slopes, and land use within the river basin). At present an unambiguous relation between river flow and input loads cannot be given; the relation will be different for each river.

When the concentration of a substance is below the detection limit, two load estimates (upper and lower values) are calculated by OSPAR and Helcom. The lower estimate is calculated such that any result below the detection limit is considered equal to zero, while for the upper estimate the value of the detection limit is used. Where this range is wide, it is an indication that most of the concentrations were below the detection limit. Previous experience has shown that in such a case the upper estimate tends to be unrealistically high (OSPAR/QSR, 2000). The detection limits depend both on the analytical method used and on the laboratory. Large differences in detection limits for the same substance are observed between countries in the Baltic Sea area (Helcom, 1998).

3.2.4. Availability and reliability of data on atmospheric inputs

Air transport models and/or extrapolation methods are used to derive atmospheric inputs into marine and coastal waters. The results differ depending on the estimation method chosen. The atmospheric input values may be biased. Therefore these values should not be compared directly with direct and riverine inputs. However, time series of atmospheric input data calculated with the same estimation method may be used for trend analysis, as the relative reliability of the data when compared to each other is not sensitive to the bias.

Results from calculating atmospheric inputs using a long-range air transport model and air emission data differ considerably from atmospheric inputs based on deposition measurements in coastal stations. The differences are due to overestimation of the measured wet deposition and underestimation of the emission data (OSPAR/ASMO 1998).

OSPAR/ASMO (1998) has data available on yearly atmospheric inputs of cadmium, mercury and lead into the North Sea (area of 525 000 km^2). These input data are estimated based on deposition measurements in coastal stations surrounding the North Sea for the period 1987–95. Helcom (1997) does not collect data on atmospheric inputs on a yearly basis. Medpol has some data on atmospheric inputs of hazardous substances for the western basin or even just the north-western part (UNEP/MAP, 1996). In this area of the Mediterranean Sea, atmospheric inputs account for 19 % of the total input of cadmium. For lead, zinc and PCBs values are estimated at 12 %, 21 % and 30 % respectively. Time series on atmospheric inputs are not available (UNEP/EEA, 1999).

3.2.5. Results of trend analysis for direct and riverine inputs

Baan and Groeneveld (2002) applied trend analysis on direct and riverine inputs of six hazardous substances (cadmium, mercury, lead, zinc, lindane and PCB_7) into the North-East Atlantic including the North Sea in the period 1990–98. The data collected by OSPAR (OSPAR/ASMO, 1998) were used. The input data are not complete. Input data are lacking, some time series are very irregular, while other time series are questionably unchanging. The yearly inputs were summed up for each of the six hazardous substances. The total 1990 input loads of each substance as well as river flow have been given a value of 100 % and inputs for other years are derived from this value. In Figure 16 the results are presented together with total river flow.

Less clearly than for nutrients (see Figure 3), the course of the curves for direct and riverine inputs of hazardous substances seems more or less similar to that for the total river flow, indicating that natural variations may have an effect on the inputs of hazardous substances.

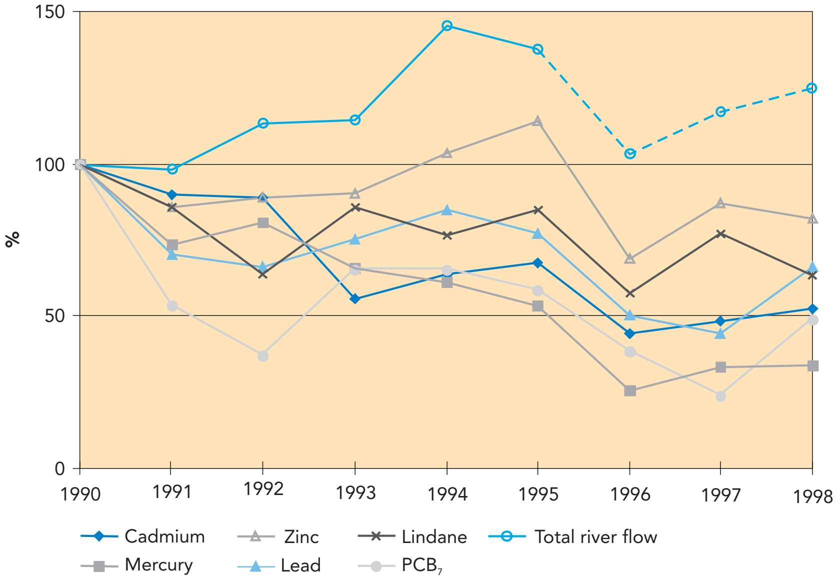
In Figure 16 the yearly total inputs of hazardous substances are divided by the yearly total river flow, yielding average river concentrations. The concentrations for 1990 have been given a value of 100 % and the values for other years are derived from this percentage.

Figure 16 and more clearly Figure 17 show decreasing trends in the inputs of hazardous substances. Among the metals the decrease in total direct and riverine input is largest for mercury and diminishes for cadmium, lead and zinc, in that order. For PCB_7 a larger decrease is observed than for lindane.

To attain a higher level of aggregation, in Figure 18 the averages of the yearly river concentrations of the six substances are presented. This yearly average may be interpreted as an (integrated) index for the input of hazardous substances. The index can be easily extended to include more substances. At such a high level of aggregation, the sensitivity of the results of trend analysis to missing data and irregularities in the data is small (Baan and Groenewegen, 2002).

Figure 16

Change in loads (sum of direct and riverine inputs) of hazardous substances into the North-East Atlantic including the North Sea in the period 1990–98, and total river flow (input of each substance and total river flow in 1990 is set at 100 %)

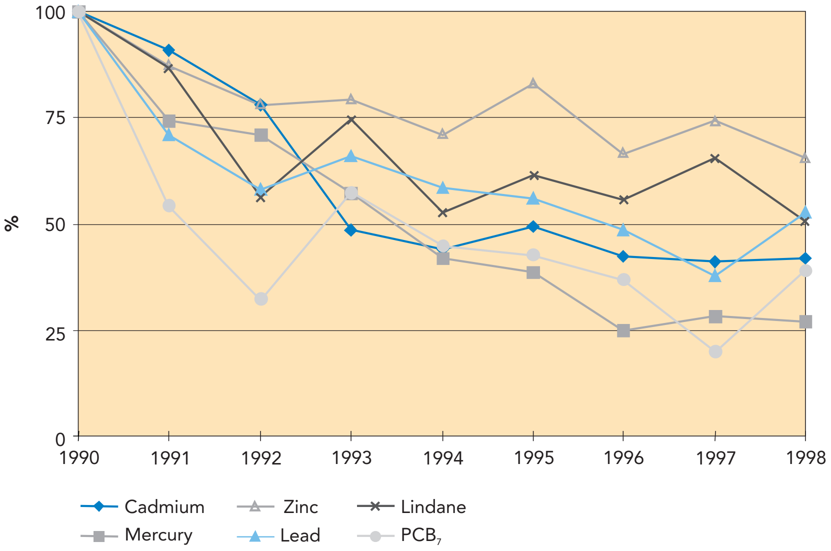


Note: Data on inputs are incomplete. Some time series are very regular, while other time series are questionably unchanging. See also note below Figure 3 on total river flow.

Figure 17

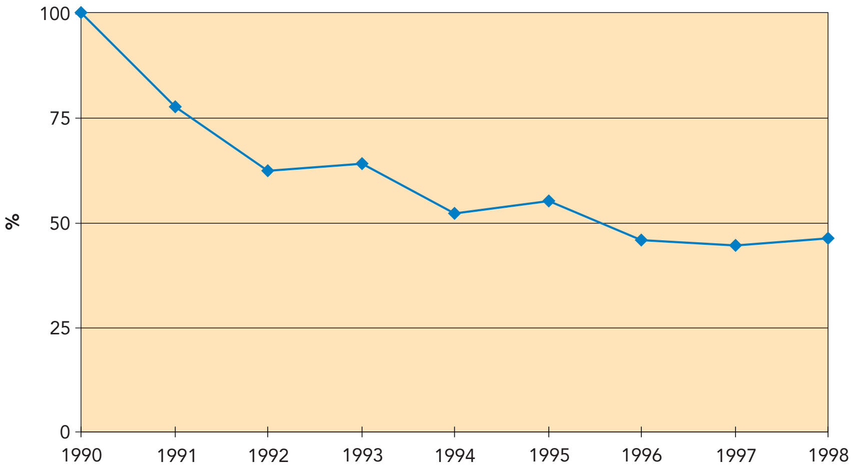
Change in average concentrations in rivers flowing into the North-East Atlantic including the North Sea in the period 1990–98 (average concentration in 1990 is 100 %)

Source: Baan and Groeneveld (2002).



Index of loads (direct and riverine input) of hazardous substances into the North-East Atlantic including the North Sea in the period 1990–98 (index value in 1990 is 100 %)

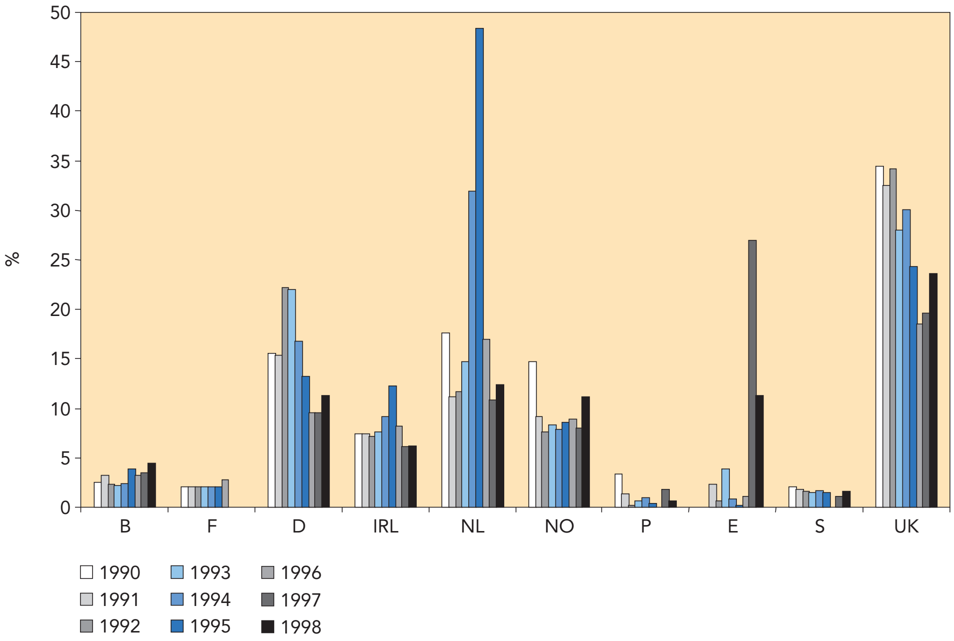
Figure 18



Contributions of countries to direct and riverine inputs of zinc into the North-East Atlantic including the North Sea in the period 1990–98 (total zinc input in 1990 is 100 %)

Figure 19

Source: Baan and Groeneveld (2002).



Note: Input data for Denmark only cover 1990 and are not presented. The whole input is assigned to the country from which the river flows into the sea. However, part of the contribution may originate from upstream countries.

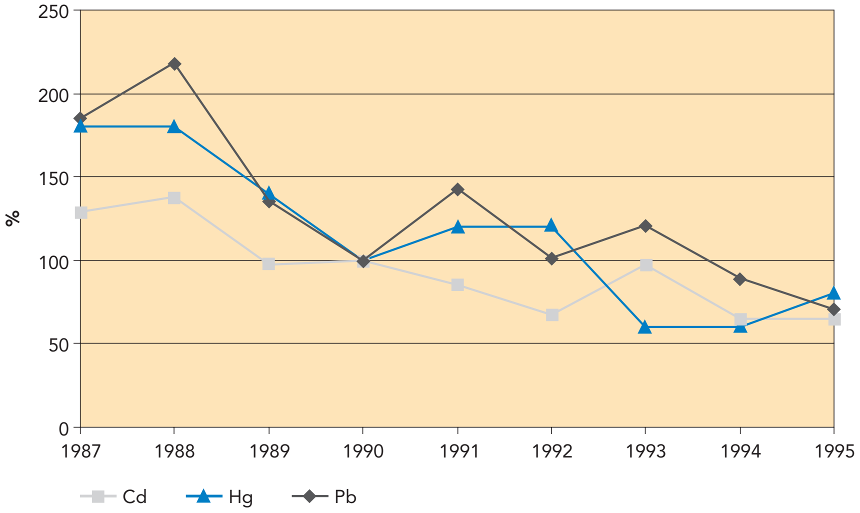
At lower levels of aggregation the results become sensitive to missing data and irregularities in the time series. The irregular course of some of the curves for concentration of hazardous substances in Figure 17 illustrates this. Results for regional seas, or at a still lower level for country contributions to inputs, are even more sensitive. However, presenting results at a low level of aggregation is a means to detect missing values and irregularities and to evoke actions to improve the completeness and quality of the data.

As an example in Figure 19 country contributions to the direct and riverine input of zinc are presented. This low aggregation level clearly reveals the reliability of the input data. Missing values are shown, as well as irregularities and questionable regularities.

Figure 20

Change in atmospheric inputs of cadmium, mercury and lead into the North Sea in the period 1987–95, based on data derived from deposition measurements in coastal stations surrounding the North Sea (atmospheric input of each metal in 1990 is 100 %)

Source: OSPAR/ASMO (1998).



3.2.6. Results of trend analysis for atmospheric inputs

Figure 20 gives an overview of the trends in atmospheric inputs of the heavy metals cadmium, mercury and lead into the North Sea in the period 1987–95. Natural variations (rainfall) may affect the yearly atmospheric inputs, but this effect is not analysed.

From Figure 20 it becomes clear that the atmospheric inputs of cadmium, mercury and lead decrease significantly between 1987 and 1995. The inputs of mercury and lead relatively decrease more in this period than the input of cadmium.

3.2.7. Usefulness of inputs of hazardous substances as pressure indicator

The input indicator provides an insight into the development of inputs of hazardous substances over time. At the highest level of aggregation (the index for inputs of hazardous substances), the sensitivity of the results of trend analysis to missing data and irregularities in the time series is rather small. Descending to lower levels of aggregation means that the results of trend analysis become more sensitive. Presenting results at low levels of aggregation reveals missing data and irregularities and is a mean to provoke improvements.

3.2.8. Recommendations for improving the pressure indicator

The usefulness of direct and riverine inputs of hazardous substances as an indicator improves significantly when the data set on inputs: (i) is made complete; and (ii) is checked on the reliability and comparability of the data. Therefore, priority should be given to this improvement. Only then should application and improvement of statistical methods and techniques to detect trends get attention.

The relation between river flow and riverine input loads should be analysed in more detail making it possible to determine normalised input loads standardised for yearly variable river flow. For rivers crossing estuaries net inputs (corrected for retention activity) should be used instead of gross inputs.

The index for inputs of hazardous substances may be easily extended to include more substances. To be able to relate index values to European policies aimed at emission abatement, one or more selections of hazardous substances should be made to be included in the index.

Guidelines for monitoring exist (OSPAR/RID, 1998; Helcom, 1998) but improvement remains possible and harmonisation on a European scale is recommended. UNEP/EEA (1999) reports that development of an effective, common Mediterranean monitoring system of measurements of contaminants and their effects is still missing, although monitoring in the Mediterranean has been in place for a long time. UNEP/EEA (1999) also states that in the Mediterranean quality assurance and control procedures should be further developed and implemented to ensure data quality and reliability. ICES (2000) indicates that efforts to standardise data collection and dissemination need to be encouraged and enhanced and that there is a need for improved, integrated monitoring through coordination and harmonisation of existing national and international monitoring activities, as well as through implementation of new methods and technology.

Simulation models are often used to determine atmospheric inputs, which make the atmospheric input data comparable to each other but often not directly to direct and riverine inputs. Further improvement of methods to determine atmospheric inputs and to integrate these inputs geographically with direct and riverine inputs is recommended.

3.3. State

3.3.1. Hazardous substances in the marine and coastal water environment

Hazardous substances will be found in the different phases of the marine and coastal water environment:

- in the water phase. Depending on the solubility the major or the minor part of a substance may be dissolved in the water phase. The other part is bound to particulate matter. Except for filter feeders, organisms take in substances from the water phase, which means that the concentration in the water phase determines the bio-availability. Homeostatic mechanisms may also regulate the concentration in the cells of organisms;
- in the sediment. The adsorption behaviour of a substance determines which part of a substance present is adsorbed on suspended solids. Equilibrium exists between the concentration of a substance in the water phase (dissolved) and on suspended solids (adsorbed). Concentrations in the top layer of the sediment are also in equilibrium with the water phase, but due to slow mixing of sediment and water the concentrations in sediment lag behind (changing) concentrations in the water phase;
- in biota. The concentration of a substance in an organism depends on the bio-availability of that substance (and thus the concentration in the water phase) and on the bio-accumulation factor for that organism and substance. The concentration in organisms will lag behind the concentration in the water phase. The time lag depends on the metabolism of the substance in an organism (intake, accumulation, degradation, excretion).

3.3.2. Availability and reliability of data on concentrations in the water phase

The concentrations of hazardous substances in the water phase in marine and coastal waters are not measured by OSPAR (Nygaard et al., 2001). In the Helcom Convention area, data are collected on concentrations in the water phase by some countries, but these data were not analysed.

3.3.3. Availability and reliability of data on concentrations in sediment

ETC/MCE composed a database with data on concentrations of hazardous substances in sediment of European coastal waters (see Nygaard et al., 2001). From an examination of the data on cadmium Baan and Groeneveld (2002) concluded that due to doubts about the reliability and comparability of the data they could not be used as state indicators.

3.3.4. Availability and reliability of data on concentrations in organisms

The data set of OSPAR/MON (1998) contains data on concentrations of hazardous substances in organisms in the North-East Atlantic including the North Sea. Quality assurance was applied on the data submitted by the countries rejecting data of ‘poor’ quality. The time series are far from complete; data of many years are lacking.

The Marinebase database (Nygaard et al., 2001) also contains data of Medpol on concentrations in organisms in the Mediterranean. Only recently have the Medpol data been scrutinised, cleared and classified according to their reliability (UNEP/EEA, 1999). For the Baltic Sea no data were available for this testing of indicators. Nygaard et al. (2001) present an overview of all data available on concentrations in organisms.

3.3.5. Results of trend analysis on concentrations in organisms

Baan and Groeneveld (2002) chose the blue mussel for indicator testing, also as this organism is widespread in European coastal waters. Data of OSPAR/MON (1998) are used, containing data on concentrations of cadmium, mercury, lead, zinc and lindane in the blue mussel in the North-East Atlantic including the North Sea in the period 1990–96. PCB_7 is excluded due to the small

Figure 21 **Change in concentrations of hazardous substances in the blue mussel in the North-East Atlantic including the North Sea in the period 1990–96 (concentration in 1990 is 100 %)**

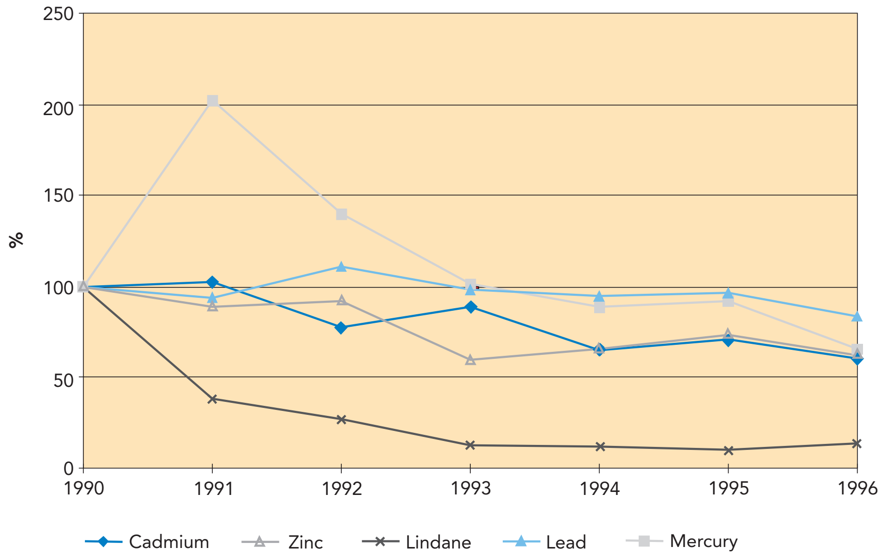


Table 1 **Upper limit of proposed EAC/BRC ranges for hazardous substances in the blue mussel**

Source: OSPAR/MON (1998).

Substance	**Upper limit of EAC/BRC ranges (*)**	**Unit**
Cadmium	550	ng/g dry weight
Mercury	50	ng/g dry weight
Lead	950	ng/g dry weight
Zinc	150 000	ng/g dry weight
Lindane	40	ng/g fat weight

(*) The lower limit is about 10 times smaller.

number of data. For each of the five hazardous substances the average yearly value of the median concentrations reported for all locations is calculated. The average concentration in 1990 has been given a value of 100 % and the average concentrations for other years are related to this value.

Results are presented in Figure 21. Lindane and mercury (from 1991 onwards) show strongly decreasing trends in the period 1990–96. For cadmium and zinc decreasing trends are also found in the period 1990–96, but the rate of decrease is less. Lead shows hardly any decrease.

For mercury, the pattern of the average yearly values of the median concentrations over time is affected by a high value in 1991 (in the Soerfjord in Norway). When analysing the data for mercury, it turns out that the high yearly values for some locations mainly determine the average values. By using median values instead of yearly averages, such extreme values do not distort the general trend. The yearly median concentrations of mercury remain about equal over time.

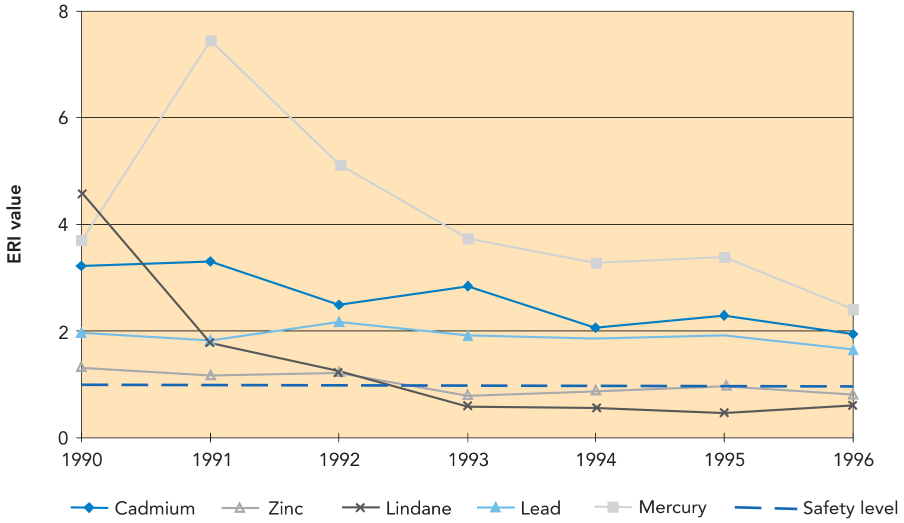
Using Medpol data on cadmium concentrations in the mussel in the Mediterranean during the period 1981–90 in the Marinebase, a trend could not be found. Since the Medpol data on concentrations of hazardous substances in organisms for the Mediterranean in the Marinebase are rather old, and their quality not assured, Baan and Groenewegen (2002) did not apply further testing on the Medpol data.

3.3.6. Method applied to determine ecological risks of concentrations in organisms

Figure 21 provides an insight into the development of the concentrations of hazardous substances in the blue mussel over time, but if and to what extent this involves an ecological risk is not shown. To gain insight into this matter, Baan and Groeneveld (2002) divided the average concentrations in the blue mussel by the upper limit of the proposed eco-toxicological assessment criteria (EAC)/background reference concentrations (BRCs) for each substance (see Table 1). The results are presented in Figure 22. The factor determined represents an ecological risk or a potential for environmental effects and is named the ecological reference index (ERI).

From Figure 22 it becomes clear that the average ERI values are highest for mercury (amounting to a value of about 7 in 1991 and decreasing after that). Concentrations below an ERI value of 1 may

Change in ecological risk from hazardous substances in the North-East Atlantic including the North Sea in the period 1990–96 Figure 22



Note: The ecological reference index (ERI) is the ratio between the concentration in an organism and the ecological assessment criteria (EAC)/background reference concentration (BRC). EAC/BRC is a range due to uncertainty in safety factors used for its calculation. The upper limit of the range has been used to calculate ERI values. Below an ERI value of 1 there is no immediate risk to marine organisms. Proposed EAC/BRC values for the blue mussel are used that have no legal significance.

be considered ecologically relatively safe. In 1993 the average values for zinc and lindane drop below this limit. For the other three substances this safety level is not reached in the period 1990–96.

Principally, assuming no synergetic or antagonistic effects of combinations of hazardous substances in species, it is allowed to add up the ERI values for the substances considered. Then the question arises how to deal with concentrations below the upper limit of the EAC/BRC ranges (ERI < 1). Adding up ERI values for different substances each below a value of 1 may result in a sum ERI > 1, suggesting that ecological risk increases significantly. We are not sure if this is true, and hence we cannot decide on: (a) adding up the ERI values calculated for each substance fully; or (b) adding up only the parts of the ERI values that are above a value of 1.

3.3.7. Usefulness of state indicator

Concentrations of hazardous substances in the blue mussel, expressed in ecological reference index (ERI) values seem a meaningful indicator. The first results with this indicator look promising and the possibilities to use this indicator should be further explored. Completeness of the data and quality assurance needs full attention.

Concentrations of hazardous substances in sediment seem less promising for use as a state indicator. The concentrations in sediments are strongly determined by morphodynamics. Sediments build up in sedimentation areas. In highly erosive areas hardly any sediment settles. This has consequences for the geographical spreading of the concentrations of hazardous substances in marine sediments and the development over time following the changes in inputs. This means that sampling sites have to be selected carefully, and that the statistical power of these concentrations for use in trend detection is probably low.

3.3.8. Recommendations for improving state indicator

Probably, other organisms than the blue mussel might also have indicator potential. Research on conditions to be fulfilled, for example sampling locations in relation to migration patterns, is recommended. Monitoring of organisms without proven indicator potential should preferably be ended to preserve time and budget for monitoring of organisms that seem more rewarding.

OSPAR (1994) and Helcom (1998) have guidelines on monitoring of hazardous substances in biota. However, it is recommended to develop and harmonise monitoring of concentrations of hazardous substances in biota on a European scale. In developing a European strategy, the results of research on determining promising indicator organisms should be incorporated. Also the geographical scale of collecting data on concentrations in biota should be agreed upon as well as the time scale (once every two or three years seems enough).

It is recommended to reconsider the monitoring of sediment concentrations. The statistical power for use in trend detection may be low. When monitoring is continued, as for concentrations in biota, it is recommended to develop and harmonise the monitoring on a European scale, addressing among others the selection of representative sample locations taking into account morphodynamics, and the geographical and time scale of collecting data.

In determining ecological reference index (ERI) values EAC/BRC values are used that are only proposed but not finally agreed as target or limit values. It is recommended to agree upon broadly accepted ecological safety levels, making the results of calculating ERI values more meaningful and valuable.

The aggregation question in the use of the ERI values still has to be answered. Adding up ERI values for different substances each below a value of 1 may result in a sum ERI > 1, suggesting that ecological risk increases significantly. Is adding up ERI values calculated for each substance allowed, and/ or should we add up only the parts of the ERI values that are above a value of 1? This aspect may be accounted for when concluding on the ecological safety levels.

3.4. Impacts

3.4.1. Biological effects

Due to their eco-toxicity, hazardous substances cause biological effects when present in certain concentrations in marine and coastal waters. There are an estimated 30 000 chemicals currently produced and used in volumes above one tonne. The potential risks are many, but for the vast majority we have only very limited, if any, knowledge of the risks they present to the environment (EC, 2001).

It is known that high concentrations of PCBs are correlated with reduced fertility in Baltic Sea grey, harbour and ringed seals, and that tributyltin (TBT) in harbours and marine and coastal waters leads to imposex among marine snails even at extremely low environmental concentrations (AMAP, 2000). Imposex is a phenomenon by which females develop male sexual characteristics and become sterile.

Studies on population parameters, health and reproduction capability in marine top predators such as marine mammals and birds have proven to be excellent response indicators for exposure of organic hazardous substances. Studies of the higher trophic levels imply that animals with well-developed enzymatic capabilities will also be included in monitoring programmes aiming at disclose environmental threats. The enzymatic capability is often a prerequisite for the organism to degrade fairly persistent compounds to more reactive and toxic compounds. This is a highly relevant argument if we also want to have a warning system for man as a consumer of seafood products.

3.4.2. Availability and reliability of data on biological effects

Since data on biological effects are not collected systematically and the relation with the pressure indicator (inputs of hazardous substances) is not clear for most substances, trend analysis could not be undertaken.

3.4.3. Usefulness of biological effects as impact indicator

The large number of hazardous substances makes it difficult to develop impact indicators for substances. The European Commission has already drawn up a list of 140 hazardous substances that need priority attention and risk assessments (EC, 2001). Not all of these hazardous substances can be monitored. A selection should be made of effects that can be clearly related to hazardous substances and that offer the possibilities of drawing the attention of policy-makers and the public. Imposex, a biological effect of the presence of TBT in marine and coastal waters, may offer such an opportunity.

3.5. Comparison of pressure and state indicators

The trends observed in the concentrations of hazardous substances in the blue mussel in the coastal waters of the North-East Atlantic including the North Sea in the period 1990–96 differ somewhat from the trends observed in the inputs of these substances in the same coastal waters. For instance lead concentrations in the blue mussel do not show a clear decreasing trend while the input of lead is clearly going down.

In Table 2 the results of trend analysis for the pressure variables (inputs of hazardous substances; see Figures 15 and 16) and for the state variables (concentrations of hazardous substances in the blue mussel; see Figure 20) are classified according to the decrease observed in the period 1990–96.

The differences between the ranking of lead and lindane for input loads and for concentrations in the blue mussel are striking. Input loads of lead decrease relatively more than the concentration in the blue mussel. For lindane this is reversed. Irregularities in the data on inputs may provide the explanation, resulting in biased results. Baan and Groeneveld (2002) examined the input data closely and made some corrections, leading to a substantial lower decrease of lead inputs in the period 1990–96 and a higher decrease of lindane inputs. Making these corrections leads to results of classification that are more easily compared, also making clear that both inputs (pressure) and concentrations in the blue mussel (state) may be considered useful indicators.

3.6. Responses

Measures to reduce the input of hazardous substances are being taken as a result of various initiatives at all levels (global, European, national, and regional conventions and ministerial conferences): the UN global programme of action for the protection of the marine environment against land-based activities; EU directives: the Water Framework Directive (2000/60/EC), the Mercury Discharge Directive (82/176/EEC), the Cadmium Discharge Directive (83/513/EEC), the Mercury Directive (84/156/EEC) and the Dangerous Substance Discharges Directive (86/280/EEC); the Mediterranean action plan (MAP); the 1992 Helsinki Convention on the Protection of the Marine Environment of the Baltic Sea Area; the 1998 OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic.

The target is a substantial reduction of the input of hazardous substances to coastal waters. In the OSPAR area the target is set at 50 % reduction of the emissions of several hazardous substances between 1985 and 1995 and zero emissions in 2010. The ultimate aim of the Water Framework Directive (2000/60/EC) is to achieve the elimination of priority hazardous substances and contribute to achieving concentrations in the marine environment near background values for naturally occurring substances. The proposed sixth environmental action programme of the EC (2001) states that the aim is to achieve an environment where the levels of man-made chemicals do not give rise to significant risks to, and impacts on, human health and the environment.

Classification of results of trend analysis for direct and riverine inputs of hazardous substances and for concentrations in the blue mussel according to the magnitude of the decrease observed in the period 1990–96

Table 2

Magnitude of decrease	Input loads	Concentrations in the blue mussel
Very large	1. Mercury 2. Cadmium	1. Lindane 2. Mercury
Large	3. Lead 4. Lindane	3. Cadmium/zinc
Moderate	5. Zinc	
Small or none		4. Lead

The new European pollutant emission register (EPER), provided for by the directive on integrated pollution prevention and control, will be of great importance in providing accessible and comparable environmental information on the emissions of pollutants from industrial sources. The EPER is a further step towards improving public awareness and towards a public 'right-to-know' about industrial pollution. It is a first step towards the development of a more fully integrated pollutant release and transfer register as called for under the Aarhus Convention on Access to Information and Public Participation on Environmental Matters.

Diffuse sources also need attention. Atmospheric pollution is caused by energy production, traffic and waste burning. Use of pesticides and fertiliser containing heavy metals in agriculture, as well as corrosion of building materials, contribute to water pollution.

Response indicators may be developed based on the reduction of emissions of industrial sources and on other land-based sources, and of materials used in buildings, etc. that may cause pollution.

3.7. Options for further development

The options for further development are partly similar to those for eutrophication (see paragraph 2.7). A better understanding of the natural loads of hazardous substances in rivers, as well as natural variations in these loads related to variations in river flow, will enable us to compare the actual state of the marine environment with the natural one and to classify coastal waters as the Water Framework Directive prescribes.

Further improvement of methods to determine anthropogenic and natural atmospheric inputs of hazardous substances and to integrate these inputs geographically with direct and riverine inputs of hazardous substances is recommended.

European policies are directed at the land-based activities that contribute to the river loads of hazardous substances. More insight into the contributions of land-based sources to the direct and riverine inputs is needed. Then response indicators may be developed based on the reduction of emissions of the land-based source.

Other organisms than the blue mussel might also have indicator potential. Concentrations in blue mussels are a coastal indicator, while concentrations in fish might be a marine indicator for hazardous substances. Research on conditions to be fulfilled, for example selection of sampling locations in relation to migration patterns, is recommended. For a few substances biological impacts are known that may be monitored directly.

4. Indicators for oil spills

4.1. Introduction

4.1.1. Policy questions

In the field of water protection and management, the Dangerous Substances Directive 76/464/EEC includes targets on oil pollution with reference to persistent and non-persistent mineral oils and hydrocarbons of petroleum origin. Targets are total elimination for persistent compounds and specific quality objectives set by EU Member States for non-persistent compounds.

The recent Directive 2000/59/EC of 27 November 2000 on port reception facilities for ship-generated waste and cargo residues aims at ensuring a major reduction in marine pollution by the provision of adequate waste reception facilities in all EU ports.

In the field of maritime safety, Directives 93/75/EEC and 95/21/EC were issued in particular to support the Marpol 73/78 Convention established by the International Maritime Organization (IMO) for the prevention of pollution from ships. Oil tankers are permitted to discharge oil or oily mixtures at the rate of 30 litres per nautical mile, but discharges are prohibited in 'special areas':

- Mediterranean Sea,
- Baltic Sea,
- Red Sea,
- Gulf of Aden,
- Antarctic,
- North Sea and its approaches,
- Irish Sea and its approaches,
- Celtic Sea,
- English Channel and its approaches, and
- part of the North-East Atlantic immediately to the west of Ireland.

In accordance with the Marpol 73/78 Convention established by the IMO for the prevention of pollution from ships, aerial surveillance operations are regularly conducted, allowing a permanent control of observed illegal slicks in the 'special areas' where discharges are prohibited.

Under the Bonn Agreement, North Sea States execute surveillance as an aid to detect and combat pollution and to prevent violations of anti-pollution regulations. The Helcom Convention established an aerial surveillance over the Baltic for the same objective. Nine Helcom member countries participate. France and Italy operate aerial surveillance planes in their Mediterranean areas of responsibility. Greece and Turkey are in the process of purchasing such planes. In the framework of the Barcelona Convention the Regional Marine Pollution Emergency Response Centre for the Mediterranean Sea (Rempec) it is intended to elaborate plans for aerial surveillance in the Mediterranean.

Improvement measures have been proposed by the EC, after the December 1999 Erika oil spill in France. Two sets of proposals were made by the Commission in March and December 2000, with the view to amend previous directives and regulations and issue a directive and two regulations for:

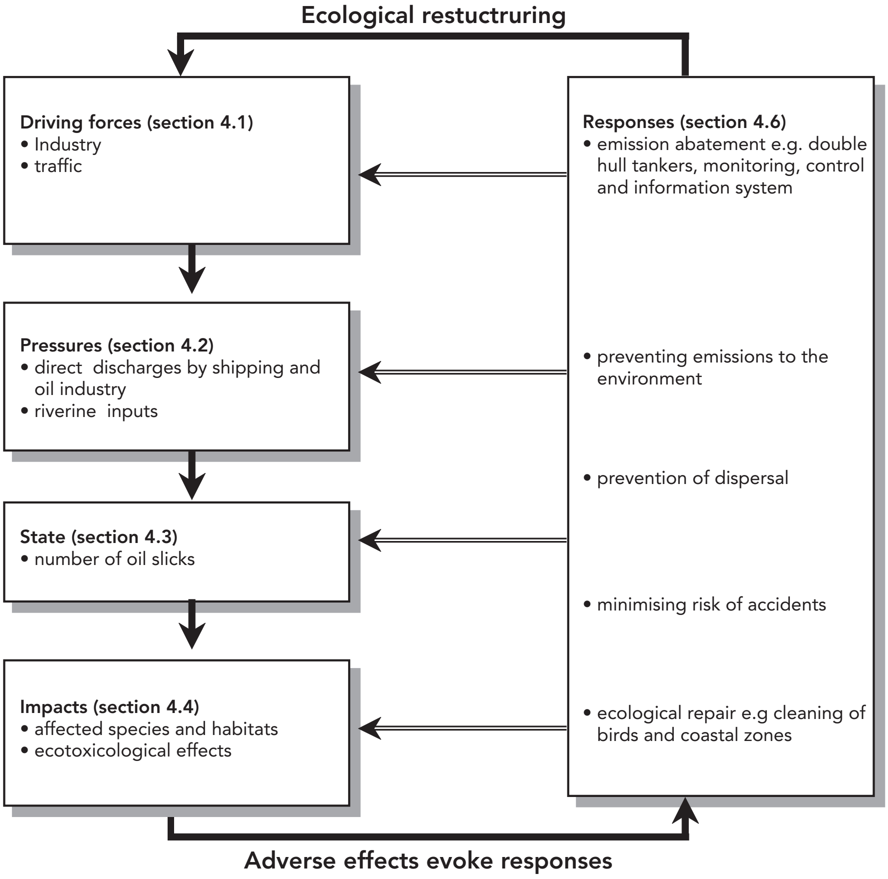
- generalising the ban on single-hull tankers;
- establishing a Community monitoring, control and information system for maritime traffic;
- establishing a fund for the compensation of oil pollution damage in European waters; and
- establishing a European maritime safety agency.

OSPAR and Helcom Conventions set targets on oil pollution from land-based sources and offshore installations, but clearly refer to Marpol 73/78 when considering oil pollution from ships.

4.1.2. DPSIR assessment framework for oil spills

The DPSIR assessment framework for oil spills is presented in Figure 23. Demand for energy and transport form the main driving forces. The two main sources of oil spills in the marine and coastal water environment are navigation and offshore oil production. The total amount of oil spilled (direct discharges into marine and coastal waters) represents the pressure, and the average area of oil slicks at the surface represents the state. These oil slicks affect sea birds (impacts) and may pollute coastal areas (see also Irvine, 2000).

Figure 23 **DPSIR assessment framework for oil spills**



Policies aim at decreasing the pressures by preventing oil spills (no or limited discharge of oil from ships) and reducing risk of oil spills by applying double hull ships. In the long run, navigation and oil production at sea should be technologically improved, preventing discharge of (waste) oil at sea and minimising the risk of accidents by applying new navigation technologies (using satellites for example).

In the mean time, oil slicks detected (state level) that may form a threat to sea birds and the coastal environment may be prevented from spreading and oil may be removed from the surface, when technically possible. Responses at the impact level include cleaning-up of coastal areas.

4.2. Pressures

4.2.1. Sources of oil spills

The main sources of oil spills in the marine and coastal environment are direct discharges from shipping and offshore oil production. River water flowing into coastal waters may also be contaminated with oil.

4.2.2. Availability and reliability of data on oil spills

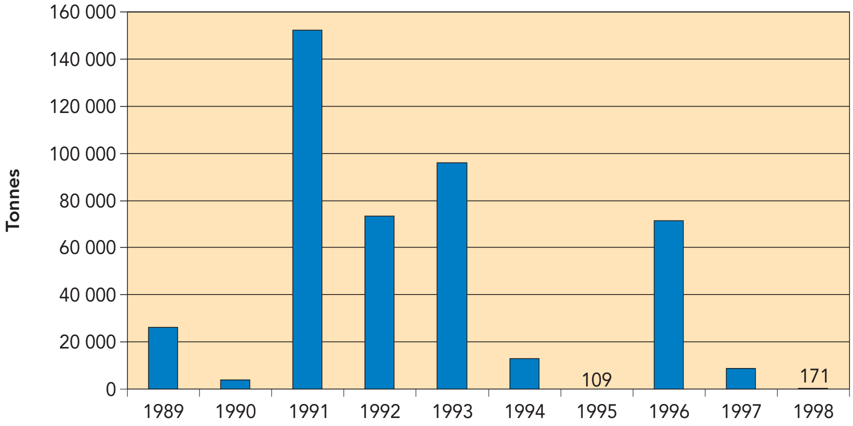
Two major sources of accidental oil spill information exist: the *Oil spill intelligence report* (*OSIR*, a commercial US news bulletin) and the International Tanker Owners Pollution Federation (ITOPF, a technical adviser). ITOPF is specialised in maritime transport spills. Since 1974, ITOPF has maintained a database of oil spills on a world scale. Spills are categorised by size (< 7 tonnes, 7–700 tonnes and > 700 tonnes). Data on spills of the class below 7 tonnes are less reliable. Though 83 % of the nearly 10 000 incidents is of the class below 7 tonnes, the contribution to the total quantity of oil spilled into the marine environment is relatively small. Therefore these data are omitted. A few very large accidents are responsible for a high percentage of the oil spilled, meaning that the figures for a particular year may be distorted by a single accident.

4.2.3. Results of trend analysis for oil spills

The majority (70 %) of oil spills above 7 tonnes in the World are caused by collisions, groundings or

Total amount of accidental oil spilled in marine and coastal waters of the 15 EU Member States in the period 1985–98 (only spills above 7 tonnes are accounted for)

Figure 24



loading/discharging operations (see Table 3). It is noticeable that a few very large accidents are responsible for a high percentage of the oil spilled. For example, in the period 1990–99 of the 346 spills over 7 tonnes, totalling 830 000 tonnes, only just over 1 % of the incidents were responsible for 75 % of oil spilled.

The yearly mean number of oil spills above 7 tonnes per year over the world has been estimated at 24.1 for the 1970–79 decade, at 8.8 for the 1980–89 decade and at 7.3 for the 1990–99 decade, showing a clear decrease over this 30-year period.

The change in the total amount of oil spilled in marine waters of the 15 EU Member States in the period 1989–98 is presented in Figure 24. The differences between the years are very large. The smallest amount of oil spilled is observed for the year 1995, and the largest amount for 1991 (corresponding to 1 400 times the smallest amount).

4.2.4. Usefulness of oil spills as pressure indicator

Accidents and accidental oil spills are largely responsible for the total amount of oil spilled in marine waters. Consequently, the total amount of oil spilled in European marine waters changes strongly between years, making it difficult to relate the yearly amounts to efforts put into pollution abatement and improvement of navigation safety.

Measures should reduce the risk of oil spills. This will lead to decreasing amounts of oil spills. Due to the large differences between the years longer time periods are needed to determine decreasing trends in the amounts of oil spilled in marine waters. For instance 10-year averages may be used. Then oil spills will become a very useful pressure indicator.

Incidence of oil spills above 7 tonnes by cause in the period 1997–98

Table 3

Incidence (%) by cause	
Operations	
Loading/discharging	23.0
Bunkering	1.8
Other operations	3.5
Accidents	
Collisions	23.8
Groundings	22.1
Hull failure	8.6
Fires and explosions	2.6
Other	14.6

Source: Eurostat, based on data from ITOPF.

4.3. State

4.3.1. State variable

Oil spills in marine and coastal waters lead to oil slicks. The number of oil slicks observed as well as the surface area covered may be considered state variables.

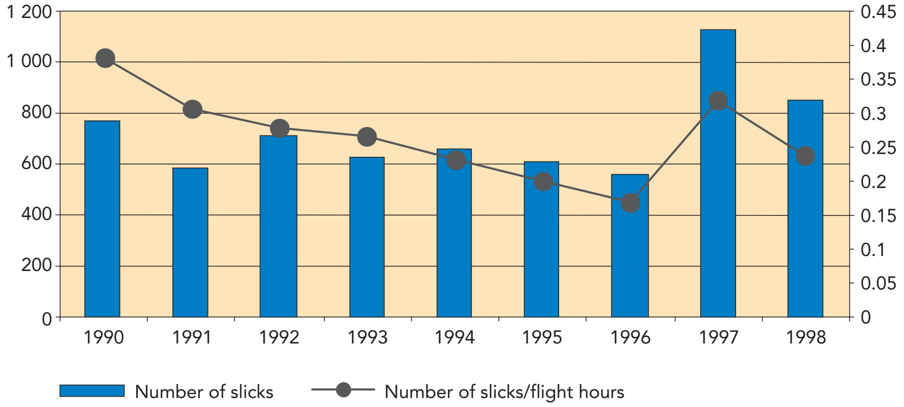
4.3.2. Availability and reliability of data on oil slicks

Under the Bonn Agreement, eight North Sea States execute aerial surveillance as an aid to detect and abate pollution and to prevent violations of anti-pollution regulations. Eight countries participate: Belgium, Denmark, France, Germany, the Netherlands, Sweden, the United Kingdom, and Norway. The Helcom Convention established an aerial surveillance over the Baltic Sea for the same objective. Nine countries participate: Denmark, Germany, Finland, Sweden, Poland, Russian Federation, Latvia, Lithuania and Estonia. The flight frequency in both areas will have an effect on the number of slicks detected.

No data on oil spills from aerial surveillance are available for the North-East Atlantic Ocean. Oil spills from aerial surveillance in

Figure 25 **Number of observed oil slicks from aerial surveillance in the period 1990–98 for both the North Sea and the Baltic Sea**

Source: Bonn Agreement (for the North Sea) and Helcom (for the Baltic Sea).



the French and Italian Mediterranean areas exceed 200 occurrences per year. But the data are available only at national level and not commonly reported under the Barcelona Convention. There is no other report for the Mediterranean Sea, where there are about 40 oil-related sites (pipeline terminals, refineries, offshore platforms, etc.). Much of the Black Sea is severely polluted with oil, especially near ports and river mouths, probably as a result of heavy boat traffic in the Black Sea; oil pollution along shipping lanes is especially heavy and is suggested to be caused by discharges of ballast water and bilge oil.

4.3.3. Results of trend analysis for oil slicks

In Figure 25 the results of aerial surveillance in the North Sea and the Baltic Sea for the period 1992–96 are presented. Both the total number of oil slicks as well as the number of slicks observed per flight hour are presented. More flights may enforce keeping the rules, thereby decreasing the number of oil slicks.

The number of oil slicks observed is declining steadily. The high frequencies in 1997 and 1998 correspond to a methodological artefact due to the reporting of very small oil spills (less than 1 m^3*) by one country.*

4.3.4. Usefulness of oil slicks as state indicator

The number of oil slicks observed may be considered a very useful indicator, both for the state as well as for responses. This indicator gives clear and direct results of oil pollution abatement policies. Moreover the aerial surveillance to identify the perpetrators and the number of oil slicks is an effective control measure that prevents illegal oil discharges.

4.4. Impacts

Oil spills are mainly confined to navigation corridors and may cause significant local damage to beaches, fish and shellfish, and bird populations. There are a lot more oil slicks from illegal discharges of oil than from accidents.

The biological effects of oil stranded onshore may be disproportionately large when key species or processes are affected. Clean-up and treatment techniques applied to spilled or stranded oil can also have significant, long-lasting effects and need to be carefully evaluated prior to use (Irvine, 2000).

Gathering information of impacts of oil spills on the marine and coastal environment was not a task in the project aimed at writing this report. Therefore such information is not included.

4.5. Responses

Under the Bonn Agreement, North Sea States execute surveillance as an aid to detect and combat pollution and to prevent violations of anti-pollution regulations. The Helcom Convention established an aerial surveillance over the Baltic for the same objective. France and Italy operate aerial surveillance planes in their Mediterranean areas of responsibility. Greece and Turkey are in the process of purchasing such planes. In the framework of the Barcelona Convention the Regional Marine Pollution Emergency Response Centre for the Mediterranean Sea (Rempec) it is intended to elaborate plans for aerial surveillance in the Mediterranean. These aerial surveillance operations, when conducted regularly, will allow a permanent control of observed illegal slicks in the 'special areas' where discharges are prohibited. The number of flights may be considered a useful response indicator.

Improvement measures have been proposed by the EC, after the December 1999 Erika oil spill in France. Two sets of proposals were made by the Commission in March and December 2000, with the view to amend previous directives and regulations and issue a directive and two regulations for:

- generalising the ban on single-hull tankers;
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- establishing a fund for the compensation of oil pollution damage in European waters; and
- establishing a European maritime safety agency.

The recent Directive 2000/59/EC of 27 November 2000 on port reception facilities for ship-generated waste and cargo residues aims at ensuring a major reduction in marine pollution by the provision of adequate waste reception facilities in all EU ports.

Response indicators may be developed based on the degree of implementation of these directives and the proposed improvement measures.

5. Integrated indicators

5.1. Introduction

5.1.1. Policy questions

In the preceding chapters the development of indicators for eutrophication and hazardous substances is discussed within the DPSIR framework. Parameters have been tested on their usefulness to describe for these themes the relation between human activities and the effects on the marine and coastal environment.

Apart from the need to develop indicators for themes like eutrophication and hazardous substances information is needed on the biological quality of the environment and the effect of various disturbances (multi-stress) due to human activities on this quality. Integrated (impact) indicators should be developed.

The Water Framework Directive (EU, 2000) states that information on biological quality elements is needed. By determining these elements and comparing them to type-specific conditions, marine and coastal waters may be classified (ecological status classification).

The proposed sixth environmental action programme (EC, 2001) addresses the need for healthy and balanced natural systems. Therefore, we need to know more about the state of the bio-diversity and the pressures and trends. Data are lacking severely in this area and organisations like the EEA and the national statistical and information bodies need to turn their attention to basis information gathering in this area.

5.1.2. Relevant biological quality variables

The Water Framework Directive (EU, 2000) states that information is needed on the following biological quality variables in coastal waters:

- composition and biomass of phytoplankton;
- composition and abundance of other aquatic flora;
- composition and abundance of benthic invertebrate fauna.

5.2. Biological quality

5.2.1. Availability of data

Data on phytoplankton biomass, benthic vegetation and fauna are scarce as these data are not collected systematically (Ærtebjerg et al., 2001). Some countries gather data for their coastal waters and determine these biological variables and, if possible, trends over time. Examples are Greece (Annex 2), Italy (Vollenweider et al., 1998), the Netherlands (Kabuta and Duijts, 2000) and Norway.

5.2.2. Indicators for phytoplankton

Diversity of phytoplankton

Kabuta and Duijts (2000) report the results of research done to determine indicators that may be useful in describing the ecosystem state of the North Sea for policy-making. The diversity of phytoplankton is calculated with the Shannon-Wiener index:

$$H = -\sum_i P_i \log_2 P_i$$

in which: P_i = ni/N
n_i = number of individuals of species i
N = total number of individuals.

H is summarised over all species present. The value of H is determined by the number of different species and the evenness of the distribution of species. When all species are present in equal numbers a value of 1 is calculated.

Kabuta and Duijts (2000) report that the H value of phytoplankton increased in the Dutch part of the North Sea in the period 1990–97 due to an increased evenness.

Trophic index

Vollenweider et al. (1998) developed a trophic index (TRIX) integrating chlorophyll-a, oxygen saturation, total nitrogen and phosphorus to characterise the trophic state of coastal waters. TRIX values are calculated with the formula:

$$\text{TRIX} = k/n \times \Sigma\ (M_i — L_i)/(U_i — L_i)$$

in which: k = 10 (scaling the result between 0 and 10)
n = 4 (number of variables that are integrated)
M_i = measured value of variable i
U_i = upper limit of variable i
L_i = lower limit of value i.

The resulting TRIX values are dependent on the upper and lower limit chosen. When location-specific limits are chosen (based on the natural conditions occurring), the TRIX values calculated indicate how close the current state is to the natural (ideal) state. However, comparing TRIX values of different areas becomes more difficult. When a wide, more general range is used for the limits, TRIX values for different areas are more easily compared to each other.

Phytotoxins

Several toxic or harmful phytoplankton species have been recorded. The toxins may be transmitted to humans through consumption of contaminated seafood and become a serious health threat. Thus, toxins are monitored in shellfish with the aim to protect consumers.

Five human syndromes are presently recognised to be caused by consumption of contaminated seafood. The European Union is currently focussing on three of them:

- amnesic shellfish poisoning (ASP),
- diarrhetic shellfish poisoning (DSP),
- paralytic shellfish poisoning (PSP).

An indicator has been developed based on these data on ASP, DSP and PSP (Annex 3).

In Figure 26 shellfish poisoning events are presented for the coastal waters of the 15 European Member States. The number of events fluctuates between the years, and a general trend is not observed.

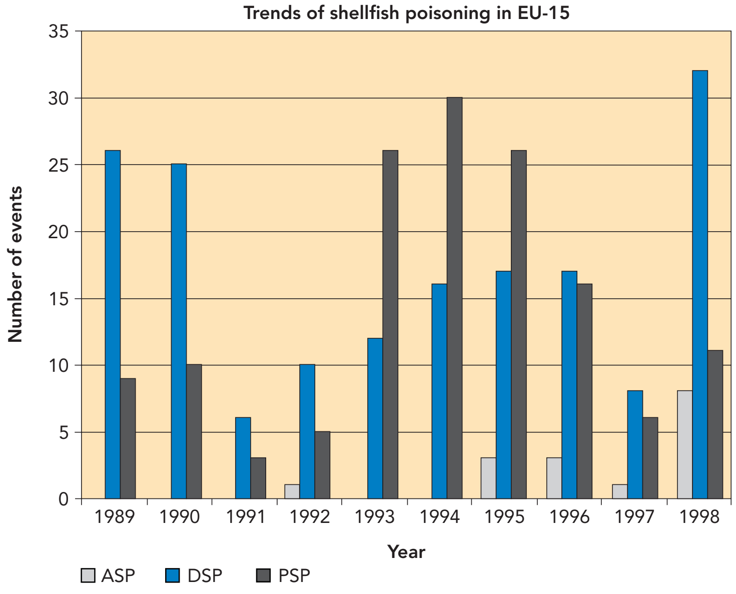
This indicator might be improved if the percentage frequency of samples collected and analysed for toxins, which show positive response for ASP, DSP and PSP respectively, could be given.

5.2.3. Indicators for aquatic flora

Aquatic flora other than phytoplankton is composed of macro-algae and sea grasses. The Water Framework Directive requests monitoring of the composition and abundance of such benthic vegetation.

Philips and Durako (2000) argue that seagrasses may be one of the best indicators of changes in coastal waters. Because most seagrasses are benthic-perennial plants, they are continuously subject to stress and disturbances that are associated with changes in water quality along the land/sea interface. Thus, seagrasses act as indicators of net changes in water quality variables which tend to exhibit rapid and wide fluctuations when measured directly.

Changes in ASP, DSP, PSP in EU-15 from 1989 to 1998 Figure 26



Source: ICES-IOC HAEDAT (harmful algae event database).

Depth distribution of macro-algae is an indicator for water quality, since with increased eutrophication the abundance of unicellular algae increase, which is reducing the light availability of bottom living macro-algae and is reducing so the ability of macro-algae to live in deeper waters.

The task of gathering data on aquatic flora or the development of an indicator were not included in the testing of indicators for this report and will be dealt with in the coming years.

5.2.4. Indicators for benthic fauna

Case study on indicators on benthic fauna in Greek coastal waters

The benthic community structure in Greek coastal waters was analysed for parameters with indicator potential (see Annex 2). Benthic community structure has been said and proved to be a reliable measure of ecosystem 'health'. Thus, monitoring of benthic ecosystems, although it may be time-consuming, has often been applied in environmental impact studies (fisheries, domestic/industrial effluent, dumping of solid waste, etc.). Historically, knowledge of the marine benthic community structure and its functioning have relied upon the most widely used benthic parameters:

- the number of species (S);
- the abundance (N) or population density expressed as number of individuals per m^2;
- the community diversity using the Shannon-Wiener index (Shannon and Weaver, 1963) or other indices;
- the ecological identity of the dominant species the so-called 'key species';
- trends in benthic parameters.

A general evolutionary pattern of the macrobenthic communities of the soft-bottomed substrate under the influence of a perturbation factor (of anthropogenic origin) has been described worldwide, based on the works of Reish, 1959; Peres and Bellan, 1973; Pearson and Rosenberg, 1978; Salen-Picard, 1983.

The changes that a benthic community undergo under the influence of the disturbance from an initial state of high diversity and richness in species and individuals are as follows:

1. a regression of the species strictly linked to the original conditions of the environment;
2. certain tolerant species considered as pollution indicators tend to monopolise available space. A limited increase of diversity can be observed in this state. The community structure remains recognisable even if degraded (subnormal zone);
3. a destruction of the community is recorded; certain species exist and develop, apparently independently of each other. Species diversity decreases and becomes minimal (polluted zone);
4. the macrobenthos disappears (zone of maximal pollution).

Based on real values and/or ranges of values of the above parameters from the Greek seas, these variables were tested as possible indicators to assess the state of the ecosystem. The test shows that the number of benthic species S can be a reliable measure of environmental stress provided that the sampling procedure is standardised (mesh sieve size, surface area sampled, depth range, sediment type, and taxonomic level). Standard values (range of values) of S for 'normal' communities should be developed for different depths and sediment type to be used as reference values in monitoring studies. Deviation from such values will then be indicative of the degree of environmental stress.

The abundance of benthic organisms N in a given area varies greatly and cannot be used as a reliable measure of environmental stress. On the other hand, the abundance of 'key species', if well defined, might be a good indicator. Key species in the Greek classification of pollution (five classes) have been identified, looking at opportunistic species, tolerant species and sensitive species. Since key species differ between coastal waters, for each coastal water the key species with indicator potential have to be determined.

Community diversity (H) using the Shannon-Wiener index can be used to classify the benthic communities' state related to the degree of pollution:

$$H = -\sum_i P_i \log_2 P_i$$

where $P_i = n_i / N$ (n_i the number of individuals of the ith species and N the total number of individuals). High diversity values normally

Classification of pollution in fresh waters based on diversity of soft-boffom founa Table 4

		Classes				
	Indicator	**V Normal zone**	**IV Transition zone**	**III Moderately polluted**	**II Highly polluted**	**I Badly polluted**
Diversity of soft-bottom fauna	Shannon-Wiener index (H)	> 4.6	4–4.6	3–4	1.5–3	< 1.5

are correlated with high numbers of species and indicate beneficial environmental conditions.

When evaluating community diversity H, one should take into account separately its two components together with the fauna data, in order to detect extreme abundance of opportunists indicating disturbance. For instance, in transition zones between two succession stages, diversity may be high, even higher than normal, whereas the community is disturbed.

Community diversity (H) in Greek waters has been calculated to range between 1.12 to 6.81, if calculated on pooled data. Figure 27 shows the distribution of H in 116 sites all over Greece. However, if calculated on a standard sampling unit (0.1 m^2*) the maximal value is 5.76 bits/unit. Certainly community diversity is lowered by severe pollution stress compared with control areas or years. Five classes of pollution can be defined for the Greek coastal waters and a pristine area using H.*

Values lower than 1.50 bits per unit (class I) have been calculated at the badly polluted areas of the Saronikos Gulf (zone I), between 1.5 and 3 (class II) for highly polluted areas of Thermaikos and Saronikos (zone II), 3–4 (class III) for moderately polluted (zone III) areas, 4–4.6 (class IV) for transitional zones (zone IV) and over 4.6 (class V) for normal zones (see Table 4). The maximum values of H coincide with the pristine areas of Sporades marine park, Kyklades plateau, Rhodes Island, Ionian Sea and Petalioi Gulf Aegean): 6,81 bits per unit.

Local improvements in the number of (benthic) species and the Shannon-Wiener index value for benthic communities in the period 1997–99 can be found but also deterioration of the ecosystem at some other locations in the Saronikos Gulf.

It is concluded that indicators (diversity index level and key species) can be used as effective tools only when a background base is available pertaining to the specific ecosystem.

Analysis of benthic fauna in coastal waters of the Netherlands

Kabuta and Duijts (2000) report the results of research done to determine indicators that may be useful in describing the ecosystem state of the North Sea for policy-making.

Macro-zoobenthos is an important link in food-web chains, meaning that the diversity of macro-zoobenthos (determined with the Shannon-Wiener index) may be used as an indicator of the ecological quality. Oxygen deficiency may result in loss of diversity and biomass. Other human activities like bottom trawl fisheries and water pollution also affect macro-zoobenthos.

Kabuta and Duijts (2000) reported an increase in the diversity of macro-zoobenthos (Shannon-Wiener index value) in Dutch coastal waters in the period 1995–98. This was due to an increase in evenness.

The macro-zoobenthos species can be divided into: (1) suspension feeders; (2) interface feeders; (3) surface deposit feeders; and (4) subsurface deposit feeders. Based on

Distribution of Shannon-Wiener community diversity (H total, X axis) over 116 Greek sites (Y axis); average diversity (H AVG): H/0.1 m^2) Figure 27

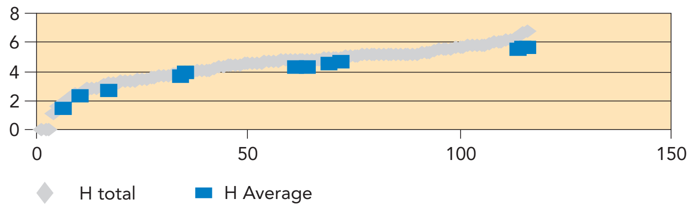


Table 5 **Classification of organic content in sediments and diversity (H) of soft-bottom fauna in Norwegian waters**

		Classes				
	Parameters	**I Very good**	**II Good**	**III Fair**	**IV Bad**	**V Bad**
Sediments	Organic carbon (mg/g)	< 20	20–27	27–34	34–41	> 41
Diversity of soft-bottom fauna	Shannon-Wiener index (H)	> 4	4–3	3–2	2–1	< 1

this division, the trophic structure of macrozoobenthos (infaunal trophic index = ITI) can be determined using the formula:

$$ITI = 100 — 100/3 \text{ x } (0n_1 + 1n_2 + 2n_3 + 3n_4)/(n_1 + n_2 + n_3 + n_4)$$

in which n_1, n_2, n_3 and n_4 are the number of individuals sampled in each of the above mentioned groups. ITI values near 100 mean that suspension feeders are dominant and that the environment is not disturbed. Near a value of 0 subsurface feeders are dominant, meaning that the environment is probably disturbed strongly due to human activities.

Kabuta and Duijts (2000) found one area north of the Wadden Sea where the infaunal trophic index value clearly decreased in the period 1991–98.

Analysis of benthic fauna in coastal waters of Norway

The benthic community structure in the inner part of the Tvedestrandsfjorden was analysed. This fjord has a restricted deep-water exchange and has been a recipient of large amounts of organic material. Oxygen deficiency near the bottom and accumulation of organic material in the sediment are important factors affecting the fauna status and gradients in the area.

The diversity of the soft-bottom fauna in the area was determined using the Shannon-Wiener formula and this diversity was compared to the organic carbon content in the sediment. Data were used from the period 1983–99. Based on the results, five ecological classes were defined (see Table 5).

5.3. Usefulness of integrated indicators

Integrated indicators like the Shannon-Wiener index (H) for the diversity of ecosystem communities, the infaunal trophic index (ITI), and the trophic index (TRIX) all require a larger monitoring effort than simple, directly to determine, variables. These integrated variables may be useful in determining the ecological state of coastal waters and probably they may be applied for trend analysis. These integrated indicators and also key species as indicators can be used as effective tools only when a background database is available pertaining to the specific ecosystem.

For the moment knowledge of and experience with the use of these variables on the basis of national monitoring and assessment programmes is limited. Analysis has been mainly done at a project basis at country or local level. Although it is yet too early for solid conclusions on the usefulness of these variables for assessing the indicator ecological quality at the European level, the results of testing in different regional seas is promising but an application at pan-European scale still needs to be developed.

6. Present state of indicator development

6.1. Introduction

To be useful, an indicator must preferably have statistical and communicative power. Statistical power means the probability that a certain change of a variable can be established with a fixed precision. Communicative power means that the meaning of the indicator is clear and that the development of the indicator over time can be presented in such a way that trends may be observed easily and can be understood and related to trends in indicators of other levels of the DPSIR framework. It also means that the indicators reflect the effects of policies.

The statistical and communicative power may be related. For instance an indicator with a relative large natural variability has less statistical power, since long time series are needed to detect trends. Then the communicative power may also be low. Data processing (smoothing of the development over time) may be necessary to make the trends clearly visible. This smoothing will improve the communicative power as well.

The present state of indicator development is discussed below by comparing the indicator potential and the progress in development made thus far.

6.2. Indicator potential

Three classes of indicator potential are distinguished:

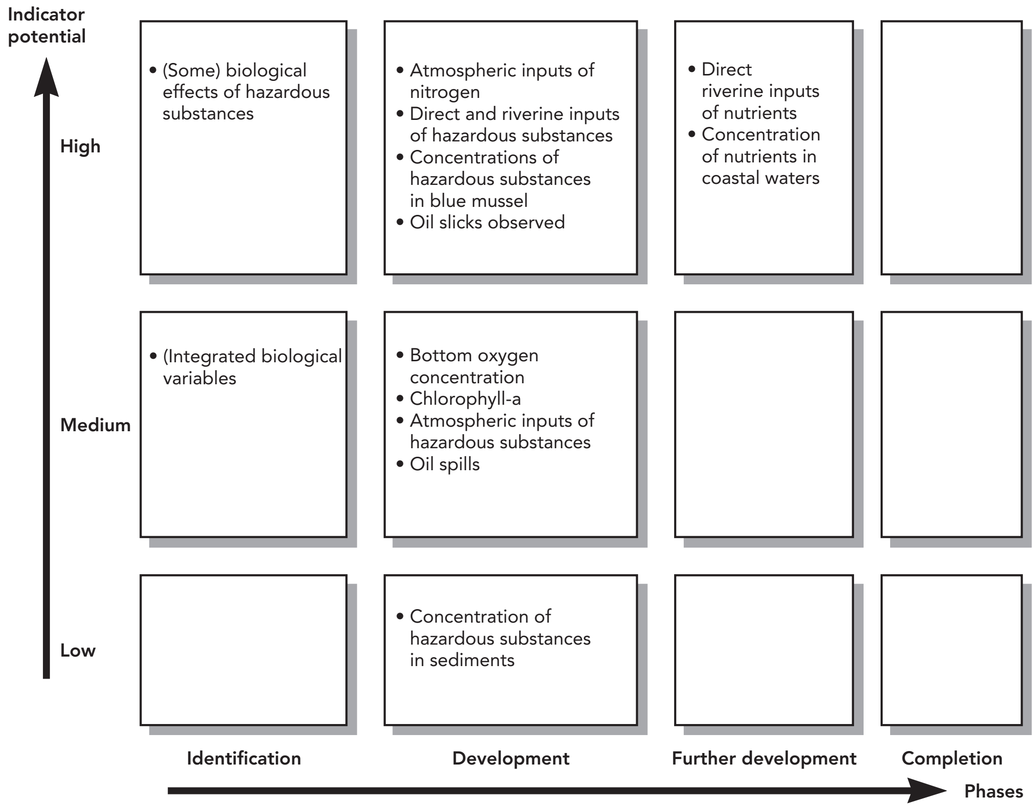
- high potential. The variables have statistical and communicative power and are useful in policy-making and evaluation on a European scale. They fit in the DPSIR framework showing clear relations with indicators at other levels of this framework. Possible trends are easy to detect with these variables. Data collection is relatively easy and quality assurance of the data poses no special problems;
- medium potential. The variables have somewhat less statistical and communicative power than high potential indicators and may be somewhat less useful in policy-making and evaluation. For instance a regional scale may be appropriate instead of a European one. The relations with indicators at other levels of this framework may be less clear. Trends are still relatively easy to detect. Data collection may become somewhat less easy and quality assurance of the data may pose some problems;
- low potential. The variables have a limited statistical and communicative power and the usefulness for policy-making and evaluation may be limited. The indicator may have value only on a local and/or regional scale. The relations with indicators at other levels of this framework are not clear. Trend detection may be difficult and data collection may become troublesome, possibly posing problems with quality assurance of the data.

In Figure 28 the parameters that are discussed in the preceding chapters are listed according to their indicator potential. A high indicator potential is assigned to the pressure indicators (inputs of nutrients and hazardous substances), and the concentration of nutrients in coastal water, hazardous substances in the blue mussel and oil slicks observed (state indicator), and some biological effects of hazardous substances (impact indicator). Concentrations of hazardous substances in sediments (state indicator) are classified as low potential. The remaining variables are considered to have medium indicator potential.

Integrated biological indicators are meant to describe the ecological state of marine and coastal waters and are not related to a specific theme. The indicator potential of these variables is not clear yet, and therefore these variables are classified as of medium potential.

The work of Sherman (2000) is relevant here. He distinguished eight types of disturbances and investigated the possibilities to relate these to a small number of disturbance indicators. These impact-based indices are meant to measure multiple marine ecological disturbances (MMEDs) and may be used to help define major impacts.

Figure 28 **Estimation of the present state of indicator development for the themes eutrophication and hazardous substances**



Indicator potential	Identification	Development	Further development	Completion
High	• (Some) biological effects of hazardous substances	• Atmospheric inputs of nitrogen • Direct and riverine inputs of hazardous substances • Concentrations of hazardous substances in blue mussel • Oil slicks observed	• Direct riverine inputs of nutrients • Concentration of nutrients in coastal waters	
Medium	• (Integrated biological variables	• Bottom oxygen concentration • Chlorophyll-a • Atmospheric inputs of hazardous substances • Oil spills		
Low		• Concentration of hazardous substances in sediments		

6.3. Progress in indicator development

With respect to the progress made in indicator development four phases are distinguished:

- identification phase. In this initial phase, the possibilities to develop an indicator are explored. Data are collected and some trials are carried out, for instance on a local scale, to find out if a variable may be useful and might have indicator potential;
- development phase. In this second phase, indicator development starts. Data are collected covering several years and for more than just a few locations. The resulting time series are used to detect trends. When the results of trend detection are promising, and data collection does not pose serious problems, the next phase may be reached;
- further development phase. In this phase the usefulness of the indicator must be tested and demonstrated. The relation with indicators at other levels of the DPSIR framework must be made clear. When policies are expected to have effects on the indicator, clear trends must be found. When considered useful the development may pass to the last phase;
- completion phase. In this last phase the focus is placed on completing data collection, both quantitative and qualitative. The geographical coverage is enlarged: all relevant marine and coastal waters must be covered. Time coverage is also improved and quality assurance of the data gets full attention making trend detection reliable.

In the two last phases special aspects and characteristics may be accounted for, like the natural variability of parameters. For instance riverine inputs vary yearly with river flows. For better comparing the input loads between years and for trend detection a thorough understanding of this variability is wanted. By using information on the relations between river flow and input loads, the loads may be standardised independent of variations in yearly river flows (see also paragraph 2.2.2). A better understanding of the natural variability and insight into the natural conditions and the present deviations from the natural state may be helpful in determining quality classes for the marine and coastal water environment, taking into account local conditions.

In Figure 28 the parameters that have been discussed in the preceding chapters are listed according to the progress made in development. None of the indicators are placed in the completion phase. Most progress is made with the development of direct and riverine inputs of nutrients (pressure indicator) and the concentrations of nutrients in coastal waters (state indicator). The major part of the variables discussed in the preceding chapters are placed in the second phase (development), meaning that a lot of work still has to be done before the development may be considered completed. All biological variables (impact indicators) except chlorophyll-a are placed in the first phase of initialisation.

6.4. State of indicator development

In Figure 28 the parameters that have been discussed in the preceding chapters are listed according to the progress made in development. Scaling against the classes of indicator potential provides an insight into the present state. It should be noted here that the assignation of the indicators to the different classes of indicator potentials and development phases is subjective. The classes of indicator potentials and phases are not clear-cut; they pass into each other rather smoothly. Therefore, the place of an indicator in the scheme in Figure 28 must not be taken as exactly true. The places are meant as indications and are open for discussion.

In (further) developing indicators, priority should be given to indicators with a high potential. However, progress can be made only when indicator development is accompanied by the development of a harmonised monitoring strategy, on a European scale. For instance, efforts are needed to standardise collection and dissemination of the data and to harmonise existing national and international monitoring activities (see also paragraph 3.2.8).

Collecting data and time series long enough to make trend detection reliable, takes a number of years. So, even for the indicators placed in the third phase of development, more years may be needed to give the indicator its full power on a European scale in policy-making and evaluation.

6.5. Need for, and possibilities of, updating and upgrading indicators

6.5.1. Need for updating and upgrading indicators

Indicators must provide an insight into the development over time of the state of the marine and coastal environment related to European policies. Presenting the results of the development should preferably be done in such a way that the period to be covered may be extended easily showing the development in the years added. Extending the time period to be covered means that annual/regular updates of the data set are needed.

When the geographical coverage of the data is enlarged the need may arise to incorporate that in the way the indicator is presented. On the one hand, this may mean that at an aggregated (European) level of presentation more areas need to be covered, and on the other hand separate results need to be shown for more marine and coastal waters.

Policies may change over time, for instance placing the focus on different substances. Indicators and the way in which they are presented should preferably be able to cope with that. Similar to the geographical coverage, it means that on the one hand at an aggregated level of presentation more substances need to be covered, and on the other hand separate results need to be shown for specific substances.

6.5.2. Flexibility to adapt indicators

The notion 'flexibility' is used to describe the possibilities to adapt the indicators and the way they are presented to possible future needs. The highest flexibility is realised when the indicator and the way the results are presented for that indicator can be easily adapted to a different setting. For instance, the geographical coverage may be increased, time series may be lengthened, and (hazardous) substances and indicator species may be added.

Determining and discussing the flexibility of indicators only makes sense for indicators for which trends are detected (see the figures in this report). In Table 6 the flexibility of these indicators is presented. Since the way of presentation often determines the ease to adapt to another setting, the flexibility is given at the level of the figures showing trends in the preceding chapters.

Table 6 **Flexibility of indicators dependent on the way of presentation**

Indicator	Way of presentation	Flexi-bility	Ease of adaptation
Inputs of nutrients (pressure)	Figures 3 and 4; aggregated over substances and seas	High	Years may be added lengthening the curves; adding inputs of more coastal waters will change the figure (new calculations needed)
	Figure 5; aggregated over regional seas	High	Years may be added lengthening the curves; completing input data will change the figure (new calculations needed)
	Figure 6; nutrient composition aggregated for North Sea	High	Years may be added lengthening the curves; completing input data will change the figure (new calculations needed); similar figures are possible for other seas
	Figure 7; wet deposition on nitrogen into the North Sea	High	Years may be added lengthening the curves; other methods applied to determine input data will change the figure (new calculations needed); similar figures are possible for other seas
Nutrient concentrations (state)	Figures 8 and 10; phosphate and nitrate aggregated for Baltic Sea and North Sea	High	Years may be added lengthening the curves; completing data will change the figure (new calculations needed); similar figures are possible for other seas
	Figure 9; phosphate aggregated for coastal waters	Low	New calculations are needed when years are added and/or when geographical coverage changes; results are dependent on period chosen and may change significantly when adding one or more years
Bottom oxygen concentration, hypoxia and chlorophyll-a (impacts)	Figure 11; bottom oxygen aggregated for Baltic Sea and for North Sea	High	Years may be added lengthening the curves; completing data will change the figure (new calculations needed); similar figures are possible for other seas; smoothing of the time series may overcome problems interpreting results due to high natural variability
	Figure 12; bottom oxygen aggregated for coastal waters	Low	New calculations are needed when years are added and/or when geographical coverage changes; results are dependent on period chosen and may change significantly when adding one or more years
	Figures 13 and 14; hypoxia and chlorophyll-a in Danish coastal waters	Medium	Current situation and change between years is presented; new calculations are needed when years are added; similar figures are possible for other seas
Inputs of hazardous substances (pressure)	Figures 16 and 17; aggregated over seas	High	Years may be added lengthening the curves; completing data will change the figure (new calculations needed)
	Figure 18; aggregated over substances and over seas	Medium	Years may be added lengthening the curves; completing data will change the figure (new calculations needed); adding substances also changes the figure (new calculations needed)
	Figure 19; country contributions	Low	Valuable for quality assurance of data; adding years or countries means making new figures
	Figure 20; atmospheric inputs into the North Sea	High	Years may be added lengthening the curves; other methods applied to determine input data will change the figure (new calculations needed); similar figures are possible for other seas
Hazardous substances in the blue mussel (state)	Figures 21and 22: concentrations and ERI values aggregated over seas	Medium	Years may be added lengthening the curves; completing data will change the figure (new calculations needed); adding substances results in a new partial figure; changes in BRC/EAC values change the figure (new calculations needed)
Oil spills (pressure)	Figure 24; aggregated over all waters	High	Years may be added lengthening the time axis
Oil slicks (state)	Figure 25; aggregated over all marine waters	High	Years may be added lengthening the time axis; completing data will change the figure (new calculations needed)

The presentation of results showing the development from year to year is considered highly flexible. Years may be added lengthening the curves and showing the development in the years added compared to the preceding period. Presentation of results showing differences between the first and the last year of a period are classified to have low flexibility. New calculations have to be made to lengthen the time period and/or to change the geographical coverage. Adding one or more years may also affect the results significantly.

6.6. Recommendations for further indicator development

In (further) developing indicators, priority should be given to indicators with a high potential (see chapter 6.2). This includes inputs of nutrients and hazardous substances as pressure indicators, concentrations of nutrients in coastal waters and of hazardous substances in the blue mussel as state indicators, and some biological effects of hazardous substance and oil slicks observed as impact indicators.

Progress can be made only when indicator development is accompanied by the development of a harmonised monitoring strategy, on a European scale. Efforts are needed to standardise collection and dissemination of the data and to harmonise existing national and international monitoring activities.

Indicators must provide an insight into the development (improvement) over time of the state of the marine and coastal environment related to European policies to improve this state. Flexibility of the indicators meaning that they can be adjusted easily to changing settings (time and geographical coverage, integration over substances and species) should get serious attention. For instance, presenting results should preferably be done in figures showing the development from year to year. Then the period to be covered may be lengthened easily and results remain comparable.

7. Conclusions and recommendations

7.1. Potential indicators and state of progress

7.1.1. Conclusions

To be useful, an indicator should have statistical and communicative power. Statistical power means the probability that a certain change of a variable can be established with a fixed precision. Communicative power means that the meaning of the indicator is clear and that the development of the indicator over time can be presented in such a way that trends may be observed easily and can be understood. Many variables have been tested for indicator potential by comparing the statistical and communicative power. The indicator potential and the phase of development differ strongly between these variables. Based on expert judgement and the experience from three years of indicator testing, high indicator potential is given to variables that have statistical and communicative power and are useful in policy-making and evaluation on a European scale. Medium indicator potential is given to variables that have somewhat less statistical and communicative power than high potential indicators and which may be somewhat less useful in policy-making and evaluation. Low indicator potential is given to the variables that have a limited statistical and communicative power and for which the usefulness for policy-making and evaluation may be limited (see chapter 6.2).

- A high indicator potential is assigned to the pressure indicators 'inputs of nutrients' and 'hazardous substances', as well as to the state indicators 'concentration of nutrients in coastal waters', 'concentrations of hazardous substances in the blue mussel' and 'oil slicks observed', and to the impact indicator 'some biological effects of hazardous substances'. Possible trends are easy to detect with these variables. Data collection is relatively easy and quality assurance of the data poses no special problems.

- The pressure indicators 'atmospheric inputs of hazardous substances' and 'oil spills' and the impact indicators 'integrated biological variables', 'bottom oxygen concentration' and 'chlorophyll-a' are considered to have medium indicator potential, because the data availability is not European wide and limited to local or regional levels.

- The state indicator 'concentrations of hazardous substances in sediments' is classified as low indicator potential, since countries could not agree on a comparable monitoring method and the interpretation of results is difficult.

- Of the variables with high indicator potential most progress has been made with the development of 'direct and riverine inputs of nutrients' and 'concentrations of nutrients in coastal waters'. But even for these variables the geographical and time coverage is far from complete. Data on country inputs are missing and time series are incomplete or too short for trend detection. For several sea areas (Baltic Sea, Iberian coast, Mediterranean Sea) monitoring is not applied on a yearly and systematic basis. That means that more years are needed to complete the development and give the indicators their full powers on a European scale in policy-making and in evaluation.

7.1.2. Recommendations

Concerning the development of indicators for the marine and coastal environment the following recommendations are made.

- In (further) developing indicators, priority should be given to the indicators classified above with a high indicator potential.

- The quality and completeness of the datasets that are used for trend analysis should get utmost attention. The value of data in trend analysis increases considerably when the data are reliable and the time series complete.

- The time and geographical coverage of data collection for indicators with firstly high and secondly medium potential should be increased to include all European regional seas and coastal waters.

- To gain full indicator power, monitoring should be fully harmonised and standardised between countries and new methods and technology should be implemented.

- Further improvement of methods to determine atmospheric inputs and to integrate these inputs geographically with direct and riverine inputs is needed.

- The monitoring of sediment concentrations for hazardous substances should be reconsidered in view of the (low) statistical power for use in trend detection.

- Other organisms besides the blue mussel might also have indicator potential for hazardous substances. Research on conditions to be fulfilled, for example sampling locations in relation to migration patterns, is recommended.

- In determining ecological reference index (ERI) values EAC/BRC values are used that are only proposed but not finally agreed as target or limit values. It is recommended to agree upon broadly accepted ecological safety levels, making the results of calculating ERI values more meaningful and valuable.

- It would be worthwhile to develop biological effect indicators for hazardous substances that may draw the attention of policy-makers and the public. Imposex, a biological effect of the presence of TBT in marine waters, appears to be a promising candidate as well as effects on birds and mammals.

- Flexibility of the indicators, meaning that the presentation of results can be adjusted easily to changing settings (time and geographical coverage, integration over substances and species), should get serious attention by EEA and Marine Conventions.

- More research is recommended on the natural loads of nutrients and hazardous substances in rivers, as well as on natural variations in these loads related to variations in river flow. A better understanding of these parameters would make it possible to determine input loads only due to human activities and to correct riverine loads for natural variations. This will improve the value of the input indicator considerably.

- More research is recommended on the natural state of marine and coastal waters related to the presence of nutrients and hazardous substances depending on local conditions and natural variations. By comparing the actual state of the marine and coastal waters with the natural one, marine and coastal waters may be classified, making it possible to compare the different marine and coastal waters in an easy and standardised way.

- To be able to relate the effect of pollution abatement policies on input loads and effects in marine and coastal waters more information should be gathered on the contributions of land-based sources (emission indicators) to the riverine inputs of nutrients and hazardous substances. Monitoring programmes for the contributions of land-based sources to riverine input loads should be attuned to and harmonised with the monitoring programmes for riverine inputs.

- The possibilities to further integrate indicators and to develop integrated indicators that describe the ecological state of marine and coastal waters need further research and testing.

7.2. Trends detected in the state of the marine and coastal environment

7.2.1. Eutrophication

The following conclusions can be drawn on eutrophication in marine and coastal waters.

- *The direct and riverine inputs of nitrogen into the North-East Atlantic including the North Sea appear to have increased slightly in the period 1990–98, while the direct and riverine inputs of phosphorus appear to have decreased slightly in this period.*

- *The composition of the nutrient load entering the coastal waters of the North-East Atlantic including the North Sea with river flow changed in the period 1990–98. The N/P molar ratio increased, the nitrate share in the input of nitrogen also increased, while the phosphate share in the input of phosphorus decreased.*

- *Atmospheric inputs contribute significantly to the total inputs of nitrogen, when considering more than a narrow area of coastal waters. A temporal change in the atmospheric inputs of nitrogen into the North Sea is not observed in the period 1987–95.*

- *The average winter concentrations of phosphate in the Baltic Sea and the North Sea decreased considerably in the period 1985–97. The average winter concentrations of nitrate also decreased in the Baltic Sea in the period 1985–97. In the North Sea the nitrate concentration varied strongly in this period. A slightly downwards trend may be observed. The N/P molar ratio increased in the coastal waters of the Baltic Sea and the North Sea.*

- *Autumn bottom oxygen concentrations in the North Sea seem to have improved from 1993 onwards. In the Baltic Sea some improvement seems to occur from 1994 onwards.*

- *Summer chlorophyll-a concentrations in European coastal waters do not show a clear trend.*

7.2.2. Hazardous substances

The following conclusions can be drawn on the inputs of hazardous substances into marine and coastal waters and the state of these waters.

- *In the period 1990–98 a decreasing trend is found in the direct and riverine inputs of hazardous substances (cadmium, mercury, lead, zinc, lindane and* PCB_7*). Among the metals the decrease is largest for mercury and diminishes in the order of cadmium, lead and zinc. For* PCB_7 *a larger reduction percentage than for lindane is observed.*

- *The atmospheric inputs of cadmium, mercury and lead decreased significantly between 1987 and 1995. The inputs of mercury and lead decreased relatively more in this period than the input of cadmium.*

- *Concentrations of hazardous substances below an average environmental reference index (ERI) value of 1 may be considered ecologically relatively safe. In the period 1990–96 in the North-East Atlantic including the North Sea, the ERI values were highest for mercury, amounting to a value of about 8 in 1991 and decreasing after that. For cadmium, zinc and lindane decreasing trends are also found. Lead does not show a clear trend. In 1993 the average values for zinc and lindane drop below the risk level of ERI value = 1. For the other three substances (cadmium, mercury and lead) this safety level is not reached in the period 1990–96.*

- *At higher levels of aggregation the results of trend detection are less sensitive to missing data and irregularities in the data than at lower levels.*

7.2.3. Oil spills

The conclusions for oil spills in the marine and coastal waters and the state of these waters are as follows.

- *The total amount of oil spilled in accidents in European marine waters varies considerably between years. In the period 1989–98 no trend is found in the yearly amount of oil spilled. For trend detection longer time series are needed.*

- *The number of oil slicks observed declined steadily in the period 1990–98. More observation flights may enforce abiding by the rules, thereby decreasing the number of oil slicks.*

7.2.4. Integrated indicators

Integrated indicators may be useful in determining the ecological state of marine and coastal waters and they can probably be applied for trend analysis with some confidence. These indicators are used only on a local and regional scale. At the local scale some trends are found.

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European Environment Agency

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Testing of indicators for the marine and coastal environment in Europe

Part 3: Present state and development of indicators for eutrophication, hazardous substances, oil and ecological quality

Annexes

European Environment Agency

Contents

Annex 1

Oxygen and chlorophyll-a concentrations as eutrophication indicators — Feasibility study

Summary

Two indicators of eutrophication have been developed and assessed using data from the Danish national monitoring programme. The first indicator describes the frequency of hypoxic conditions, which is defined as oxygen concentrations below 2 ml/l. Hypoxic conditions have severe consequences for the marine ecosystem. The second indicator describes the mean chlorophyll-a concentration in the summer period. High chlorophyll-a concentrations may increase the frequency of hypoxia. Both indicators have been tested and found appropriate as eutrophication indicators. We recommend that the indicators be tested on data from other marine community waters.

1. Introduction

Eutrophication caused by excess load of anthropogenic nitrogen and phosphorus is a major problem in many European coastal areas. Increased nutrient loads on marine community waters have had repercussions on the ecological balance of these systems (Ærtebjerg et al., 2001). The marine ecosystem is composed of a complex mosaic of interacting processes, where changes in the nutrient pressures will inevitably have consequences on the functioning of the ecosystem. Nutrient concentrations are state indicators with the closest link to nutrient pressures. Most marine waters in Europe have experienced increasing nutrient concentrations (Ærtebjerg et al., 2001). This has subsequently led to increased frequency and magnitude of algae blooms and increased risk of oxygen deficiency in bottom waters.

The objective of this study is to evaluate the usefulness of oxygen and chlorophyll-a concentrations as indicators of the state of eutrophication. This feasibility study has been carried out using data from the Danish national monitoring programme only, because the calculations required access to raw data. Aggregated data provided by ICES (Nygaard et al., 2001) could not support the indicator computations for oxygen, and only partially the computations for chlorophyll-a.

2. Oxygen indicator

2.1. Introduction

Enhanced primary production in eutrophic waters increases the sedimentation rate of organic material to the bottom. The subsequent degradation of organic material in the sediments consumes oxygen from the overlaying waters. The oxygen content in bottom waters is determined by two processes:

- the consumption of oxygen due to degradation of organic materials in the bottom water and sediments. The consumption rate depends on the amount and quality of organic material sedimenting to the bottom and on the temperature;
- the supply of oxygen from vertical mixing and horizontal transport processes. The supply rate depends on the hydrographical processes forced by wind, buoyancy and tides.

Oxygen deficiency will occur if the consumption rate exceeds the supply rate for a sufficiently long period of time for the oxygen in the bottom water to be depleted. Oxygen deficiency is only a problem in marine waters with periodic or permanent strong stratification. Marine waters can be classified into three categories.

1. Areas without oxygen deficiency. This is typically well-mixed or weakly stratified waters, where stratification is continuously broken down by tidal mixing, wind mixing, etc. or in stratified waters with a deep bottom layer regularly supplied with oxygen-rich water by horizontal advection.

2. Areas with temporary oxygen deficiency. This is typically waters with strong stratification and low horizontal advection in long periods — in general, most of the summer and autumn period. The oxygen deficiency can be created locally or transported to the area by deeper layer advection processes.

3. Areas with permanent oxygen deficiency. This is typically waters with a permanent strong stratification and low horizontal advection, which applies to many of the deep basins in the Baltic Sea and estuaries with a sill preventing the exchange of bottom water.

2.2. Indicator for oxygen deficiency

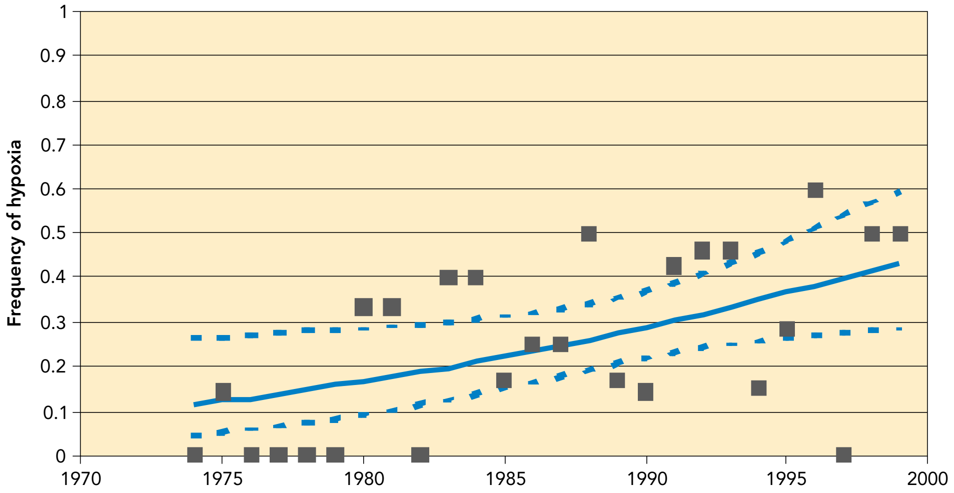
The yearly minimum oxygen concentration has become a de facto standard for describing the state of oxygen deficiency (Rosenberg, 1990). Further, minimum oxygen concentrations are frequently used in trend analyses based on the assumption of observations with equal distributions. It is recommended not to base an evaluation of oxygen deficiency on minimum concentrations, because the statistical properties of minimum values depend on the number of observations over which the minimum value has been calculated. This is intuitively easy to understand, because the more observations we have from a single year, the lower the expected minimum value will be. Thus, it is unfortunate to use the minimum oxygen concentration for state and trend analyses.

In the international literature, hypoxia is operationally defined as oxygen concentrations below 2 ml/l (Diaz and Rosenberg, 1995). It has been shown that hypoxic conditions can have detrimental effects on the benthic fauna (Diaz and Rosenberg, 1995). A better alternative to the minimum oxygen concentration is to consider the frequency of hypoxia. The frequency of hypoxia is defined as the number of observations within May to November below 2 ml/l divided by the total number of observations. This period could be determined on a regional scale when the period of potential oxygen deficiency has a different extent. The frequency describes the probability of observing hypoxic conditions from May to November. This period has been chosen as the season where hypoxia normally prevails in northern temperate waters.

A statistical method for trend analysis is to consider the number of hypoxic observations each year to be binomial distributed with parameters n and p_i, where n is the total number of observations for year i and p_i is the probability of observing hypoxic conditions in year i. The trend analysis is conducted by testing if p_i is a function of year. This method is known as logistic regression (McCullagh and Nelder, 1989). A virtue of this method is that it weights yearly frequency observations with the number of observations it has been based on. Thus, the frequency of hypoxia determined from a year with many observations of oxygen concentrations has

Frequency of hypoxia (observations marked by dots compared to the logistic regression line (solid line) with 95 % confidence limits (dashed lines))

Figure A1.1



more weight than the frequency of hypoxia determined from a year with few observations of oxygen concentrations.

The use of this indicator will be exemplified with data from a Danish station located in the southern Belt Sea between Germany and Denmark. Figure A1.1 shows the results of the analysis on this single station. The increasing trend in frequency of hypoxia is significant at a 5 % significance level (P = 0.0183). This station was deliberately chosen because it showed an increasing frequency of hypoxia.

2.3. State of oxygen deficiency

Oxygen deficiency is a significant problem in major parts of Danish waters. In the 80s the Kattegat and the Belt Sea were exposed to several events of hypoxic conditions (Andersson and Rydberg, 1988), which subsequently led to political decisions on nutrient reductions. Oxygen deficiency has also been reported in other marine community waters (e.g. Johannessen and Dahl, 1996).

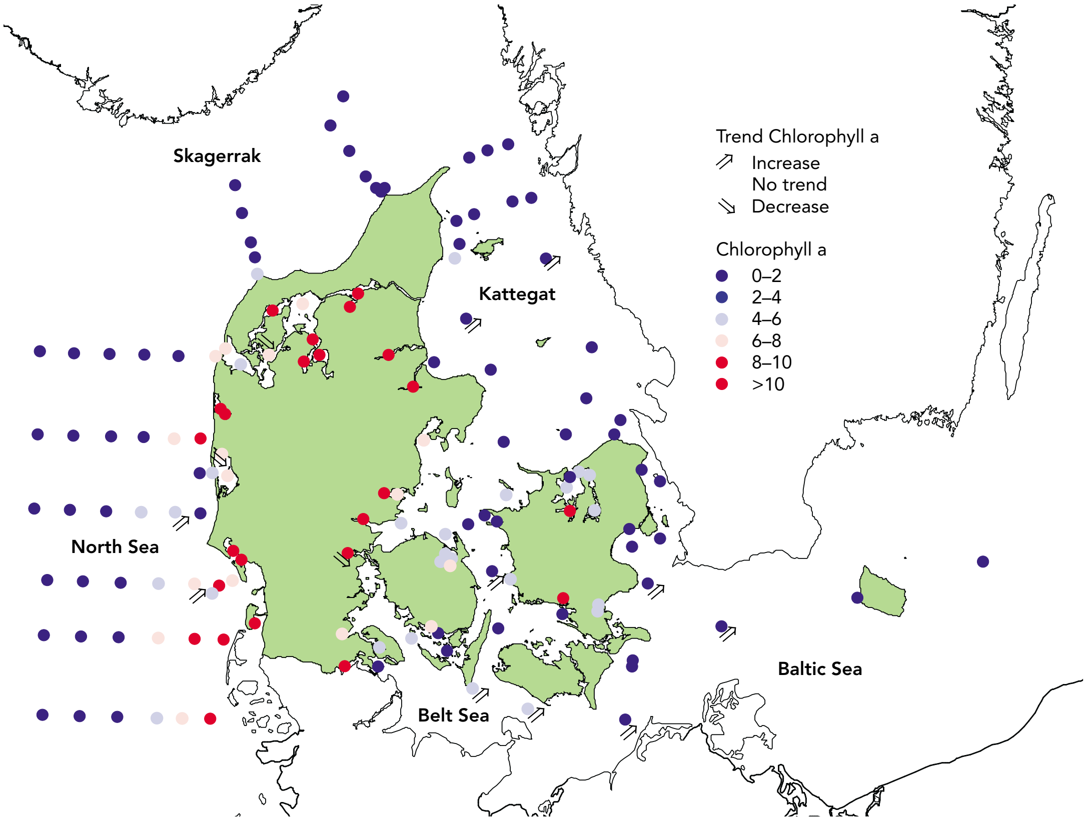
The frequency of hypoxia for the last half of the 1990s (1995 and onwards) and trends of this frequency (no specific period) are shown in Figure A1.2. It is observed that many stations are characterised as stations without oxygen deficiency problems (frequency = 0). This applies especially to all the stations in the North Sea and Skagerrak, where tidal mixing and currents induce a constant replacement of bottom waters. In Kattegat, the Belt Sea and the Baltic Sea most stations are exposed to hypoxia, and these marine areas must be considered sensitive to oxygen deficiency. Some estuaries (typically estuaries with a sill) are exposed to almost permanent hypoxic conditions. The eastern-most station in the Baltic Sea is located in one of the deep Baltic Sea basins, where almost permanent hypoxia is observed from of depths 60–70 m down to the bottom at 86 m. Almost permanent hypoxic basins are characteristic for the Baltic Sea (Helcom, 1996; Kullenberg, 1981).

The logistic regression analyses show that four out of 153 stations have a significant trend in the frequency of hypoxia (three stations show an increase and one station a decrease). Hence, there is no general trend in the frequency of hypoxia. This is due to two facts.

1. Most time series of oxygen concentration start in the 1970s or 1980s while nutrient loading had already increased significantly in the 1950s and 1960s. We can therefore not evaluate the transition from low pressures to high pressures.

2. Variations in frequency of hypoxia caused by meteorological fluctuations mask variations of lesser amplitude that accrue from changes in anthropogenic loading. Long time series are required to detect changes in the frequency of hypoxia.

Figure A1.2 **Frequency and trend of hypoxia for stations in the Danish national marine monitoring programme**



The frequency of hypoxia can be used for classification of marine community waters into areas with and without oxygen deficiency problems. In waters susceptible to oxygen deficiency, the frequency of hypoxia is a good indicator of the state of eutrophication.

3. Chlorophyll-a indicator

3.1. Introduction

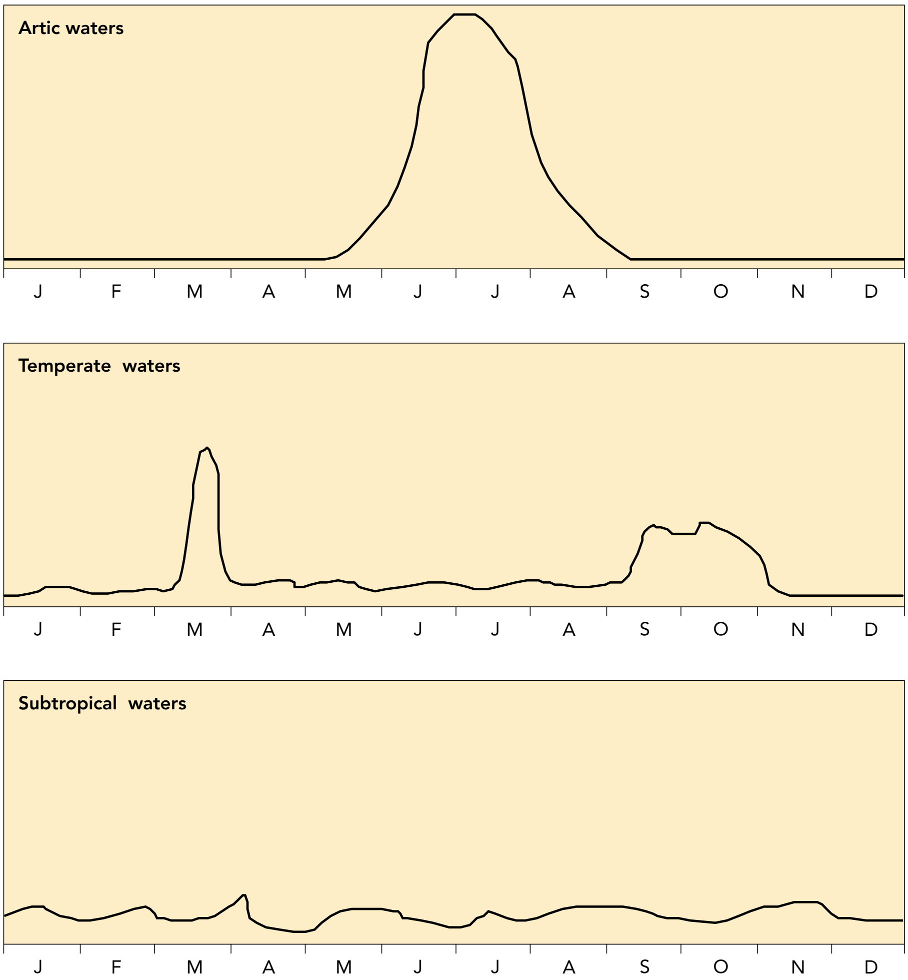
Increases in nutrient loading have enhanced primary production in marine community waters (Richardson and Heilmann, 1995); this has the potential to raise the standing stock of phytoplankton organisms. The amount of phytoplankton in the water column is governed by:

- the primary production rate, which depends on the phytoplankton biomass and is limited by the availability of nutrients and light. The phytoplankton community responds quickly to changes in the limiting factors (< 1 day). Primary production consists of new production fuelled by external supply of nutrients from land, atmosphere or deep water mixing, and regenerated primary production fuelled by recycling of nutrients from the loss processes;
- the loss of phytoplankton by pelagic and benthic grazing, sedimentation, decay and advective transport processes. Most of the nutrients in the loss of phytoplankton are recycled and used for regenerated primary production.

The response to enhanced primary production could potentially be an increase in phytoplankton biomass, if grazing on the phytoplankton community is not too high.

Chlorophyll-a is the green plant pigment used for photosynthesis, which is present in all autotrophic phytoplankton organisms. Chlorophyll-a is an adopted measurement technique for monitoring the amount of algae. The content of chlorophyll-a typically accounts for 1/40-60 of the total carbon biomass in the algae, but this ratio varies considerably depending on species composition of the plankton, and on the physiological state of the algae.

Figure A1.3 **Illustration of the seasonal variation in phytoplankton biomass and chlorophyll-a (adopted from Lalli and Parson, 1993)**



European community waters stretch from Arctic waters in the north over temperate waters to subtropical waters in the Mediterranean Sea and off the Iberian coast. The seasonal variation in phytoplankton biomass depends on the climatic conditions. This is illustrated in Figure A1.3.

Arctic waters have a characteristic unimodal seasonal variation with an intense summer bloom where light conditions are optimal. Temperate waters have a characteristic bimodal seasonal variation with a short intensive spring bloom, a summer period where phytoplankton is mainly controlled by nutrient limitation and grazers, and an autumn bloom where wind mixing events bring nutrients to the surface waters. Subtropical waters have a weak seasonal variation, where phytoplankton biomass does not change much over the year.

3.2. Indicator for chlorophyll-a

In Ærtebjerg et al. (2001) the median summer chlorophyll-a (0–10 m in April to September) was used as an indicator and a weak positive correlation to winter nitrate concentration was found. In temperate waters the period April–September corresponds to the stationary period between the spring and autumn bloom for most years. However, in colder regions of Europe, this period will include a late spring bloom or the unimodal summer bloom characteristic for Arctic waters. As a result, such an indicator may potentially yield higher summer chlorophyll-a values in the oligotrophic waters in the northern-most parts of Europe compared to the eutrophic waters in central Europe.

Another issue is the depth definition used for chlorophyll-a indicators. Hitherto, the chlorophyll-a indicator has been calculated on averaged values from 0 to 10 m. This depth definition is suitable for the North Sea and the Baltic Sea, but in the Mediterranean Sea the productive photic zone stretches further down in the water column. Thus, the median summer chlorophyll-a (0–10 m) from April to September is an inappropriate indicator of eutrophication when applied to all community marine waters.

Phytoplankton responds quickly to changes in the limiting factors, i.e. nutrients and light. In periods with nutrient limitation and no light limitation the chlorophyll concentration is partly determined by the supply of nutrients; therefore the chlorophyll-a concentration in this period will provide a good indicator for eutrophication.

We propose to use the average chlorophyll-a concentration in the period with nutrient limitation and no light limitation including observations at depths above the pyknocline as a eutrophication indicator. This requires the depth and the time interval to be defined on a regional scale. The depth integration of chlorophyll-a is chosen according to the typical stratification depth of the region, where 0–10 is considered appropriate for the North Sea and Baltic Sea. In the Mediterranean Sea a depth integration of for example 0–50 m may be considered appropriate. The time period from April to September is an appropriate period for the North Sea and southern Baltic Sea, while this period is shorter in the Arctic waters and longer in subtropical waters.

The average chlorophyll-a concentration is used as an indicator instead of the median, because chlorophyll-a concentration in the nutrient limited period often has a skewed distribution. Calculating median values of the chlorophyll-a distribution will neglect the skewness, i.e. high chlorophyll-a concentrations will not affect the median of the distribution. It is, however, important also to include the information in chlorophyll-a measurements of larger magnitude in an eutrophication indicator unless the measurements are faulty. We suggest to apply Kendall's τ to the chlorophyll-a indicator for trend analysis.

3.3. State of chlorophyll-a

The chlorophyll-a levels in 1999 at stations in the national Danish monitoring programme are shown in Figure A1.4 with arrows indicating if a significant trend was detected using Kendall's t (no specific period specified). It is observed that in the estuaries the nutrient discharges from land give rise to higher chlorophyll-a concentrations. The

coastal stations have lower chlorophyll levels compared to the estuaries, but it is also observed that coastal stations in the North Sea have higher values when approaching the German Bight.

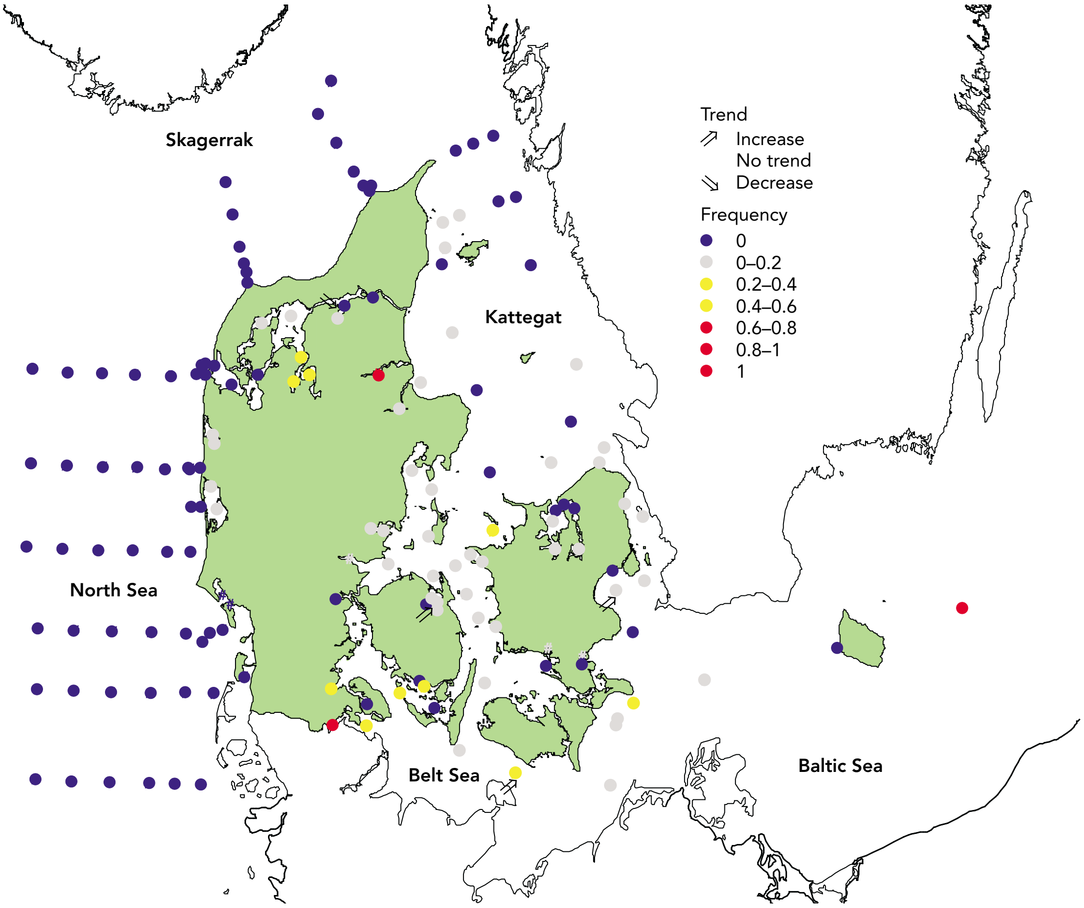
The trend analyses show that 14 out of 141 stations have a significant trend in summer chlorophyll-a concentration (10 stations show an increase and 4 stations show a decrease). Hence, there is no overall trend in the chlorophyll-a concentration, but many stations in the western Baltic Sea and Belt Sea have experienced increasing summer chlorophyll concentrations. Many of these stations have time series going back to the mid-1970s. The time series of summer chlorophyll-a for a single station is shown in Figure A1.5 revealing that most of the change in chlorophyll-a levels occurred from the 1970s to the last half of the 1980s.

Only a few of the monitoring stations in the estuaries have observations of chlorophyll-a from the 1970s, and the large change in nutrient loading occurred in the 1950s and 1960s.

The average chlorophyll-a concentration in periods with nutrient limitation and no light limitation is a good indicator for eutrophication, because the amount of plankton is related to the supply of nutrients during this period. The chlorophyll indicator also shows a consistent pattern with higher concentrations close to sources of land-based nutrient inputs to marine community waters.

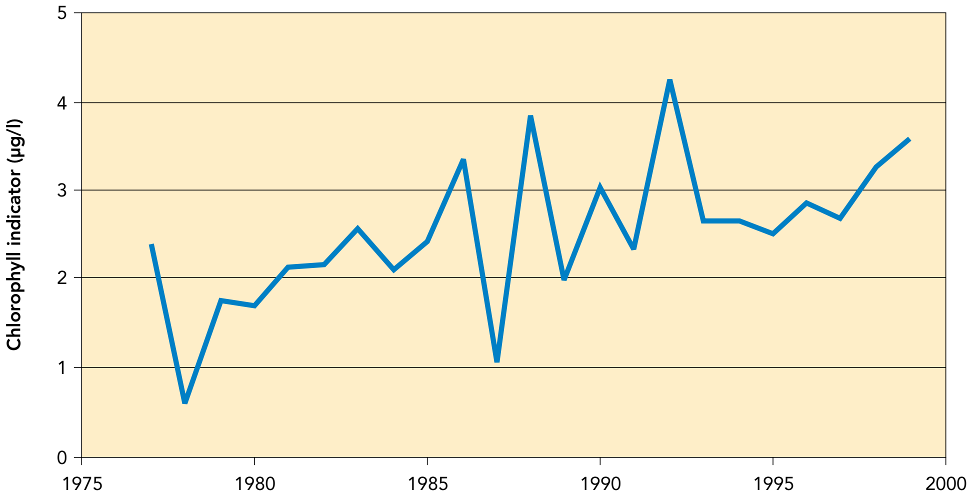
Summer chlorophyll-a concentrations for stations in the Danish national marine monitoring programme (arrows indicate significant trends analysed with Kendall's t)

Figure A1.4



Summer chlorophyll-a concentrations for Helcom Station BMP K7 in the western Baltic Sea

Figure A1.5



4. Conclusions and recommendations

The minimum oxygen concentration in the summer period has been a widely applied indicator for oxygen deficiency. This indicator is inappropriate, because it depends on the sampling frequency. Thus, an indicator for oxygen deficiency, which is not biased by the sampling frequency, is proposed. This indicator is the frequency of hypoxia, which is the proportion of bottom oxygen concentrations below 2 ml/l. Hypoxic conditions have detrimental effects on the benthic community. The proposed indicator separates marine community waters into three categories: (1) areas without oxygen deficiency problems, (2) areas with temporary oxygen deficiency problems, and (3) areas with permanent oxygen deficiency problems. In areas with temporary oxygen deficiency problems, the frequency of hypoxia provides a good indicator for the state of eutrophication, and trends in the frequency of hypoxia are analysed using a logistic regression.

Chlorophyll-a is a measure of the amount of phytoplankton. In summer periods, where the primary production is limited by nutrients only, the chlorophyll-a concentration is related to the supply of nutrients. We propose an eutrophication indicator calculated as the average chlorophyll-a concentration in a well-defined summer period, when primary production is limited by nutrients only. The chlorophyll-a concentrations are integrated over depths above the pyknocline. Appropriate intervals for the summer period and depth integration are determined according to the local conditions of the different European regions.

The diversity of monitoring data on oxygen and chlorophyll a from different marine waters in Europe should be tested for general applicability as policy relevant indicators within the EEA core set of indicators.

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Annex 2

Soft-bottom benthic indicators

1. Introduction

Soft substrata are the most widespread habitat among seabeds and support high biodiversity and key ecosystem services. The characteristics of benthic macrofauna and the associated sedimentary environment are of key importance to legislative, management and conservation objectives.

Benthic community structure has been said and proved to be a reliable measure of ecosystem 'health'. Thus, monitoring of benthic ecosystems, although it may be time-consuming, has often been applied in environmental impact studies (fisheries, domestic/industrial effluent, dumping of solid waste, etc.).

Historically, knowledge of marine benthic community structure and functioning has relied upon the most widely used benthic parameters, namely:

- number of species (S);
- abundance (N) or population density expressed as number of individuals per m^2;
- community diversity using the Shannon-Wiener index (Shannon and Weaver, 1963) or other indices;
- the ecological identity of the dominant species, the so-called 'key species';
- trends in benthic parameters.

A general evolutionary pattern of the macrobenthic biocoenosis of the soft-bottomed substrate under the influence of a perturbation factor (of anthropogenic origin) has been described worldwide, based on the works of Reish, 1959; Peres and Bellan, 1973; Pearson and Rosenberg, 1978; and Salen-Picard, 1983.

The changes that this community undergo under the influence of the disturbance from an initial state of high diversity and richness in species and individuals are as follows.

A. A regression of the species strictly linked to the original conditions of the environment.
B. Certain tolerant species considered as pollution indicators tend to monopolise available space. A limited increase of diversity can be observed in this state. The biocoenosis structure remains recognisable even if degraded (subnormal zone).
C. A destruction of the biocoenosis is recorded; certain species exist and develop, apparently independently of each other. Species diversity decreases and becomes minimal (polluted zone).
D. The macrobenthos disappears (zone of maximal pollution).

These phenomena have been described to apply for the Mediterranean (Marseilles region, specifically the coasts of the Italian peninsula: Trieste, the Bay of Elefsis, the upper part of the Saronikos Gulf, Gulf of Izmir). So a more descriptive zonation pattern has been derived from the above general scheme, applying for the Mediterranean. Three concentric zones or areas can be demonstrated based on faunistic and geochemical data (Bellan, 1985).

- Zone I (zone of maximal pollution). This zone is deprived of all animal macroscopic life and, at times, of the meiobenthos.
- Zone II (polluted zone). The population in this zone is characterised by a very small number of polychaete species. The quantitative evolution of these species can be extremely rapid.
- Zone III (subnormal zone). The subnormal zone is characterised by a notable species enrichment, but also conserves a clear dominance of polychaetes: 70–80 % of the individuals. The species of the preceding zone have practically disappeared. This zone can perhaps be divided into two subzones: an 'internal' subzone corresponding perfectly to the criteria indicated and an 'external' subzone or the 'ecotone zone' intervening before the normal zone where the action of pollution cannot be felt.

Based on real values and/or ranges of values of the above parameters from the Greek seas, they are tested as possible indicators to assess the state of the ecosystem. Finally, using as a paradigm a well-studied Greek area, a classification system is suggested where benthic 'indicators' alone would be able to 'describe' efficiently the state of the marine ecosystem.

2. Indicators tested and discussion

2.1. Indicator 'Number of species'

The number of species in a benthic assemblage varies greatly with depth. The best-known species number gradient in benthos has been: an increase of species diversity from high to low latitudes in continental shelf regions and in the open sea, a regular change from coast to abyssal plain, and an increase of species from inshore habitats to the open sea. A significant decrease in species number with depth has been observed in Greek waters (see Figure A2.1).

Grassle and Maciolec (1992) argued that coastal diversity is low compared with deep-sea and offshore habitats. In a review work Gray et al. (1997) have shown that species richness per unit area is as high, if not higher, in shallow sedimentary habitats as was reported for the deep-sea data by Grassle and Maciolec. Indeed, one of the central patterns in biodiversity, noted universally, is that the number of species increases with the area sampled. Data from the North Aegean Sea clearly show this increase with increasing sampling effort. A cumulative increase of species number with sampling effort is also exhibited in other areas. Figure A2.2, based on aggregated data collected from nine stations over three years in the Geras Gulf, shows such a trend.

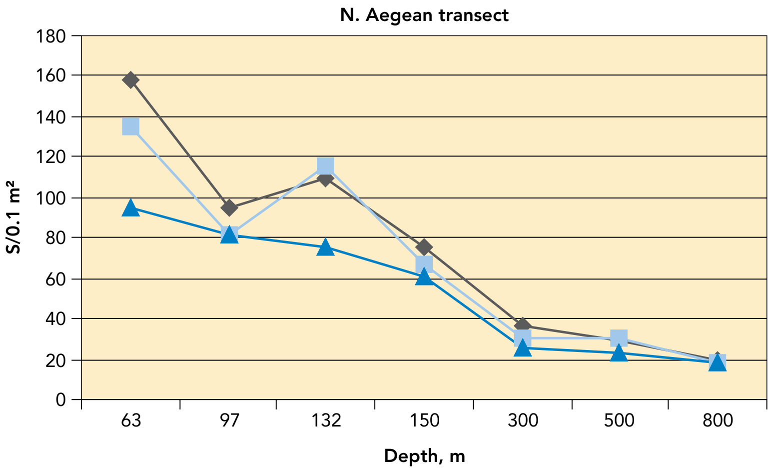
Another factor influencing directly the number of species is the sediment type. Two examples, where different sediment types hold a different species number, are presented in Figure A2.3**.** The data are from two very well studied areas of the Aegean Sea which, being away from any land-based pollution sources and thus unaffected by anthropogenic activities, serve as reference sites. In the graph, it is clear that species number per sampling unit is not dependent on season. The larger the sampling area, the higher the increase. Thus, from an average of 72.5 species /0.1 m^2, it increases to 189 species/m^2 and to 350 species/7 m^2 in the silty sand site, and from 23 species/0.1 m^2 to 78.4 species/m^2, reaching 128 species/7 m^2 in the silty site respectively.

However, when comparing the species number of a sampling unit with that of the average of either 10 samples (AVG S in figure) or of 70 samples (AVG/0.1m^2 in the legend of figure), it appears that species number of a given unit (sediment surface area) can be an accurate measure of the state of environment.

Based on data collected over a variety of soft-bottom habitats in Greek waters it appears that number of species, in undisturbed areas (reference sites), ranges between 22 and 165 species per 0.1m^2, depending on depth and type of substratum.

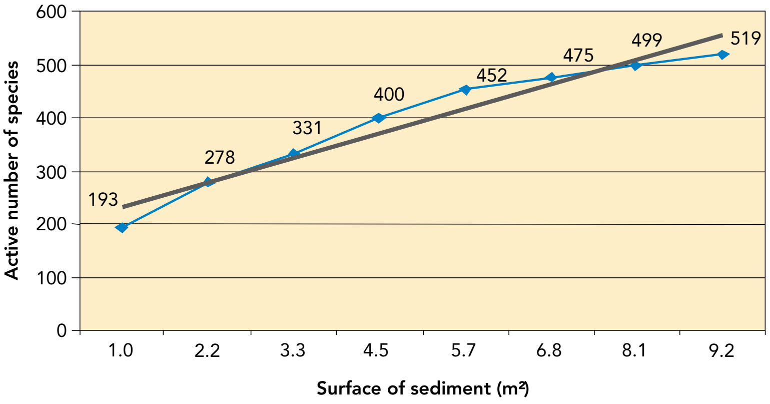
Figure A2.1 **Trend in species number with depth (coloured lines correspond to three different replicates)**

Source: Keycop project, Zenetos et al., 2000).



Increase in number of species with sampling effort

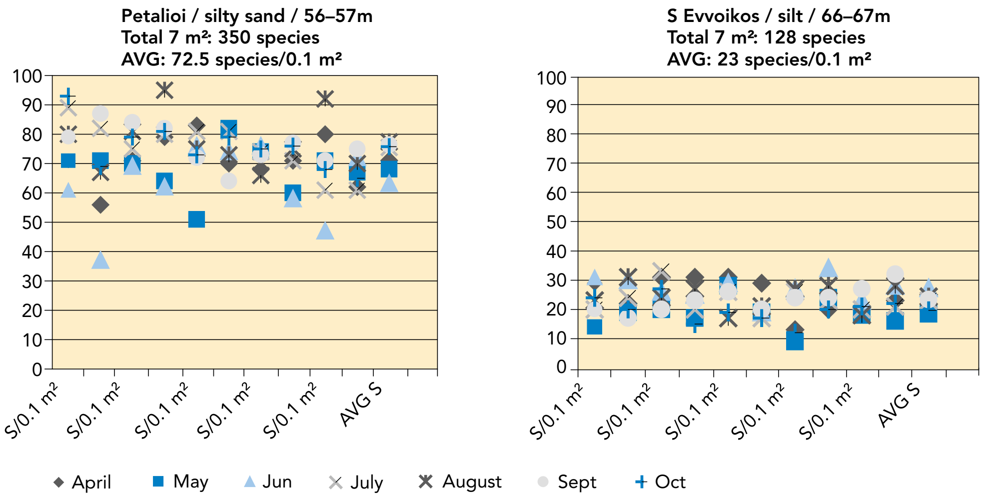
Figure A2.2



Source: Bogdanos et al., in press.

Variation in species number per sampling unit (0.1 m²) and monthly average (0.1 m²) in different habitats (depth, sediment type)

Figure A2.3



Source: TRIBE project, TRIBE, 1997.

Conclusively, the number of species (S) can be a reliable measure of environmental stress provided that it is used when comparing benthic communities:

a occurring within a well-defined sampling unit (standard 0.1 m²),from samples collected with the same gear (standard grab 0.1m², mesh sieve 0.5 mm),

b at the same depth range and sediment type (ranges to be defined per sea),

c the species identification is being done at the same taxonomic level (four major groups or all groups).

Recommendation: Standard values (range of values) of S for 'normal' communities should be developed for different depths and sediment type to be used as reference values in monitoring studies. These values, however, can be different for different seas and regions. Deviation from such values will then be indicative of the degree of environmental stress.

2.2. Indicator 'Abundance (N)'

Abundance alone, even if expressed as the same unit, which usually is the number of individuals per m², is not informative enough as a stable indicator for the state of the ecosystem. The reason is that it is too variable to show meaningful changes. A decrease with depth is a recognised trend worldwide. Data from the North Aegean Sea (Figure A2.4) demonstrate this trend. Besides depth and sediment type, abundance also depends on settlement and size of individuals. On the other hand abundance is a parameter independent of sampling effort.

Figure A2.4 **Trend in abundance with depth**

Source: Keycop project, Zenetos et al., 2000.

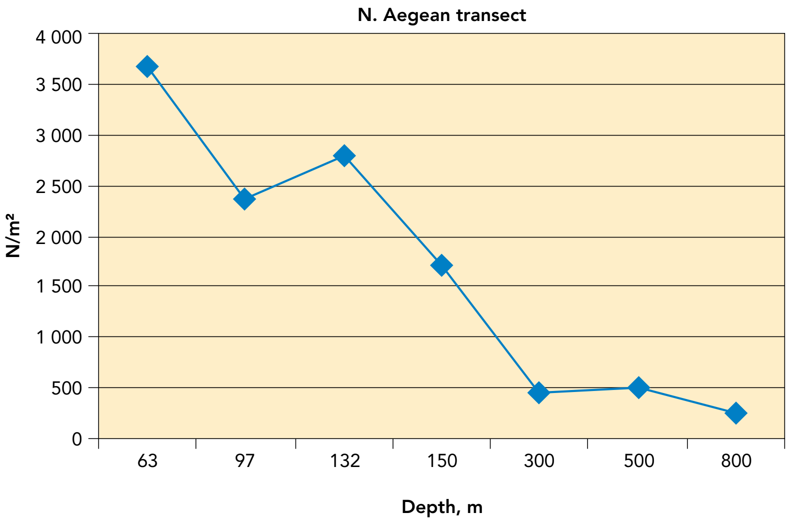


Table A2.1 **Key species, indicative of the degree of environmental disturbance**

I. Zone of maximal pollution	Azoic
II. Highly polluted zone	Opportunists: *Capitella capitata, Malacoceros fuliginosus Corbula gibba*
III. Moderately polluted zone	Opportunists: *Chaetozone sp, Polydora flava, Schistomeringos rudolphii, Polydora antennata, Cirriformia tentaculata*
IV. Transitional zone	Tolerant species: *Paralacydonia paradoxa, Protodorvillea kefersteini Protodorvillea kefersteini, Lumbrineris latreilli, Nematonereis unicornis, Thyasira flexuosa*
V. Normal zone	Sensitive species e.g. *Syllis sp.*

Abundance values in Greek waters for 'normal' ecosystems can vary from: 217 individuals/m^2 (average for Kyklades plateau; Zenetos et al., 1997) to 619 individuals/m^2 (average for Rodos area; Pancucci-Papadopoulou et al., 1999) and 1 439 individuals/m^2 (average for Sporades marine park; Simboura et al., 1995). In disturbed ecosystems, abundance can reach values as high as 8 695 individuals/m^2 (Station E7, Ionian Sea; Zenetos et al., 1997) or 4 428 individuals/m^2 (TP11, Thermaikos Gulf; NCMR technical report). These extremely high values are characteristic of very disturbed ecosystems, from either fisheries activities (the former) or from domestic and industrial pollution (the latter). In both cases, a few opportunistic species dominate at the expense of others.

Conclusively, the abundance of benthic organisms in a given area is too variable and cannot be used as a reliable measure of environmental stress. On the other hand, trends in abundance of 'key species', if well defined, would be a good indicator.

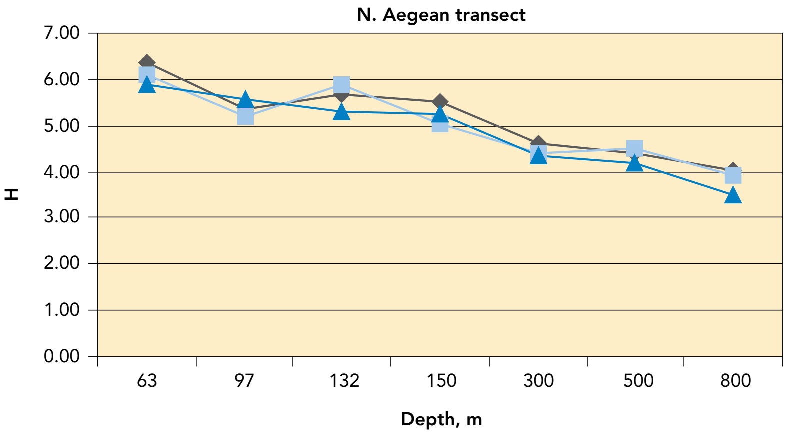
2.3. Indicator 'Key species'

Based on a synthesis of reviews on the subject and on the data presented here as case studies (Dauvin, 1993; Pearson and Rosenberg, 1978; Bellan, 1985), Table A2.1 shows the zones of pollution with the respective key species.

2.4. Indicator 'Community diversity (H)'

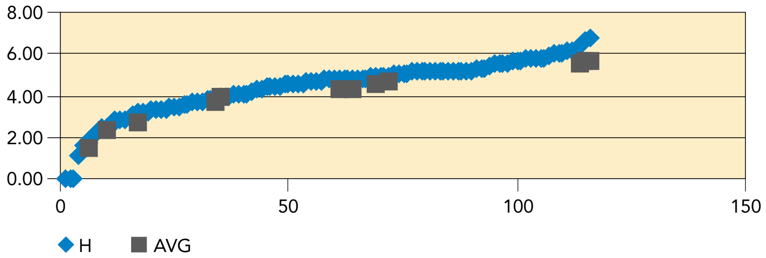
The number of species and their relative abundance can be combined into an index that shows a closer relation to other properties of the community and environment than would number of species alone. The Shannon-Wiener diversity index, developed from the information theory, has been widely used and tested in various environments. Although it reflects changes in the dominance pattern, it has been argued that it is no more sensitive than the total abundance and biomass patterns in detecting the effect of pollution and is more time-consuming. A decreasing trend with depth

Trend of community diversity with depth Figure A2.5



Source: Keycop project, Zenetos et al., 2000.

Distribution of community diversity (H) over 116 Greek sites (AVG: H/0.1 m²) Figure A2.6



Source: S. Evvoikos, Petalioi Gulfs: TRIBE, 1997; Rodos: Pancucci-Papadopoulou et al., 1999; Sporades: Simboura et al., 1995; Kalamitsi, Ionian Sea: Zenetos et al., 1997.

has been established (Figure A2.5) in Greek seas whereas increasing sampling results in minor changes of the community diversity.

Community diversity in Greek waters has been calculated to range between 1.12 and 6.81, if calculated on pooled data. However, if calculated on a standard sampling unit ($0.1m^2$) the maximal value is 5.76 bits/unit. Figure A2.6 shows the distribution of H in 116 sites all over Greece. Certainly community diversity is lowered by severe pollution stress compared with control areas or years. Values lower than 1.50 bits per unit have been calculated at the badly polluted areas of the Saronikos Gulf (zone I), between 1.5 and 3 for highly polluted areas of Thermaikos and Saronikos (zone II), 3–4 for moderately polluted (zone III) areas, 4–4.6 for transitional zones (zone IV) and over 4.6 for normal zones. The maximum values of H coincide with the pristine areas of Sporades marine park, Kyklades plateau, Rhodes Island, Ionian Sea and Petalioi Gulf Aegean: 6.81 bits per unit.

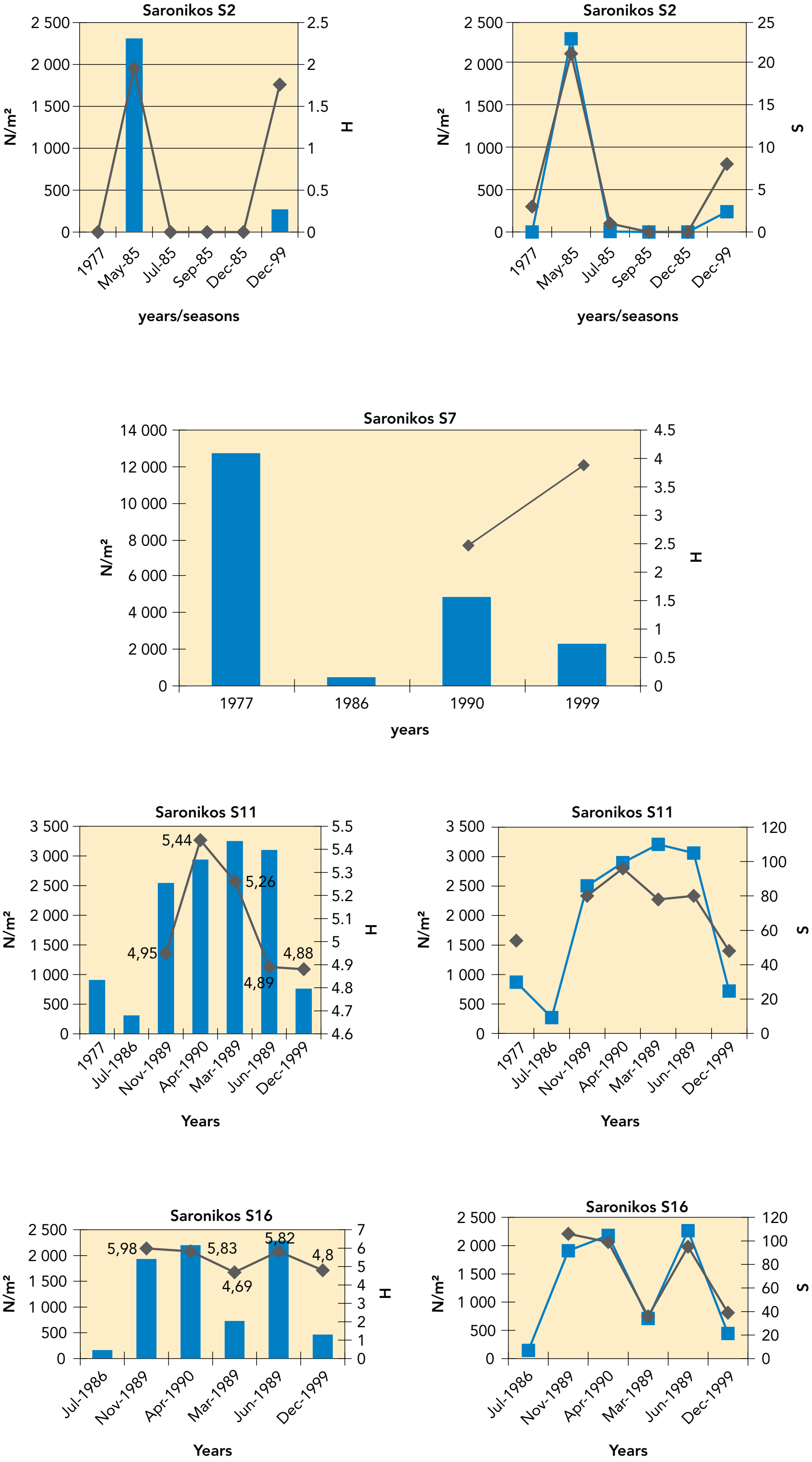
When evaluating H, one should take into account separately its two components together with the faunistic data, in order to detect extreme abundance of opportunists indicating disturbance. There are some cases where the diversity is significantly high, even higher than normal, whereas the community is disturbed. The ecotone point is a transitional zone between two successional stages after which the community returns to normal. The community at the ecotone point consists of species from both adjacent environments (enriched and less enriched). After the ecotone point the community often reaches a maximum in the number of species, probably caused by the presence of sensitive species recolonising community and tolerant species, while abundance declines to a steady state level usually found in normal communities. Thus diversity may become higher than that of the normal communities (Pearson and Rosenberg, 1978; Bellan, 1985). An example is the community at Psittalia Station S7 (Saronikos Gulf) where diversity and species number are high

Conclusively, in Greek waters based on the community diversity index alone, five classes of community health can be arbitrarily divided.

Class I: $H < 1.5$: azoic to very highly polluted
Class II: $1.5 < H < 3$: highly polluted
Class III: $3 < H < 4$: moderately polluted
Class IV: $4 < H < 4.6$: for transitional zones (zone IV)
Class V: $H > 4.6$: normal

Figure A2.7 Seasonal and inter-annual changes of diversity (H), abundance (N/m²) and species (S) at different locations (S2–S16) in Saronikos Gulf

Source: 1974 data in Zarkanellas and Bogdanos, 1977; 1985 data in Friligos and Zenetos, 1985; Zenetos and Bogdanos, 1987; 1986 data in Zenetos et al., 1990; 1989–90 data in Simboura et al.,1995; 1999 data from ETC/ Water-NCMR.



enough to classify the area to transitional zone 4. However the presence of heavy pollution indicators like *Capitella capitata* and *Malacoceros fuliginosus* reaching very high densities classifies the area to zone 2, and probably creates a deterioration of the environment causing a regression in the communities structure.

2.5. Trends in benthic community parameters

Figure A2.7 gives examples showing seasonal and inter-annual changes of the parameters discussed above in the Saronikos Gulf. It is clear that Station S2, which became azoic in the 1985 period, has been recolonised as indicated by the 1999 results. A serious improvement is also evident at Station S7, close to the sewage outfall, after the recent establishment of a treatment plan. On the other hand deterioration of the ecosystem is seen at Stations S11 and S16 situated at an increasing distance from the sewage outfall.

3. Case study: Classification of Saronikos Gulf (Gulf of Athens) based on benthic community data

If one wants to apply the above scheme to a given area, certain modifications must be made.

The benthic communities of the Saronikos Gulf, receiving the domestic and industrial effluent of Athens, have been studied since 1975. Based on data collected between 1974 and 1999, though sparse in time, all of the above zones can be recognised at least over time.

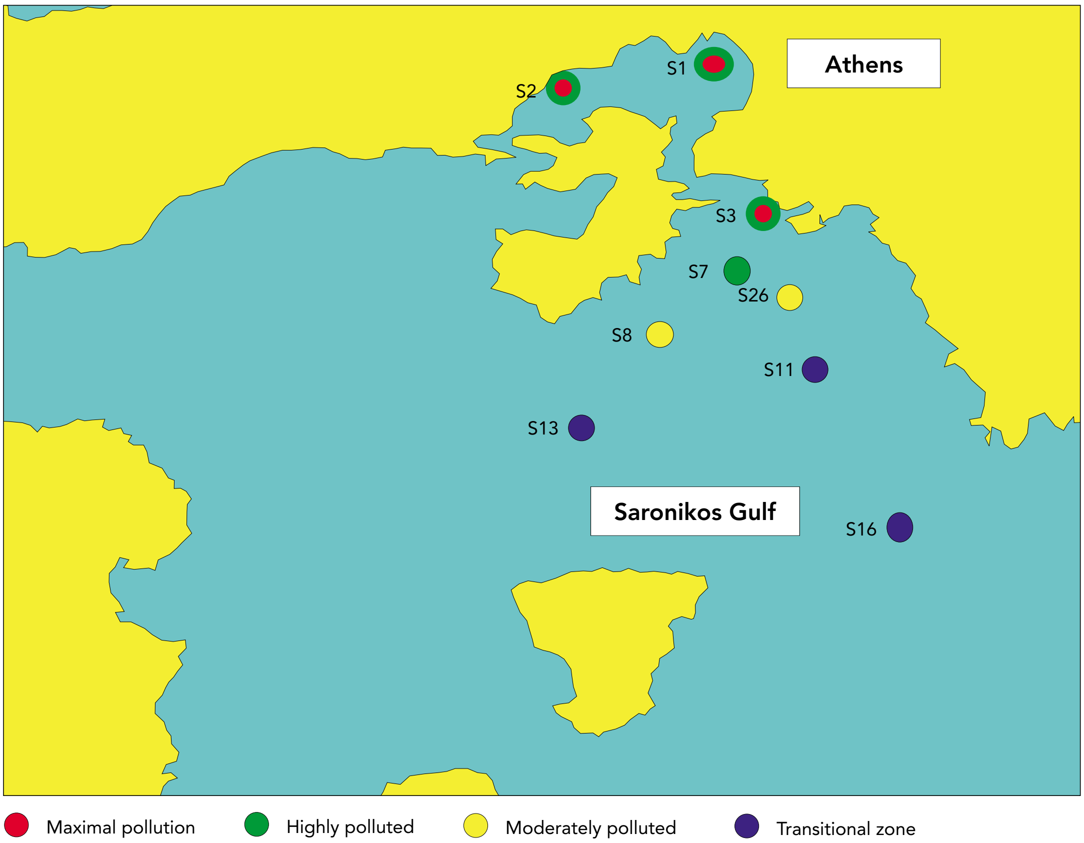
I. Zone of maximal pollution. Stations S1, S2 and S3 have been found azoic during some sampling periods in the past. Station S1 was defaunated during September 1985 sampling, Station S2 during September 1985 and December 1985, and Station S3 of Keratsini Bay was azoic in July and September 1985.

II. Highly polluted zone. Stations S1, S2, S3, S7 of the inner Saronikos Gulf and of Elefsis Bay can be classified in this zone. This classification is based on the combined criteria of the level of the communities' ecological indices and the key species. Specifically these stations are characterised by: very low average diversity ($1.5 < H < 3.89$) and average evenness ($0.55 < J < 0.86$) reflecting either very low species richness (5 species/0.1m^2, only at Station S2) or very high total densities (2 220 ind/m^2 at Station S7 and 2 070 ind/m^2 at Station S1) which indicate the presence of few heavy pollution indicators (*Capitella capitata* (S7, S2), *Malacoceros fuliginosus* (S3), *Corbula gibba* (S2)) that reach very high densities.

III. Moderately polluted zone. Stations S8 and S26 can be classified in this zone which is characterised by the following trends: still low diversity but higher than previous zone ($2.66 < H < 4.30$), higher evenness ($0.72 < J < 0.80$), higher species number ($13 < S < 41$), more moderate densities but still high ($600 < N/m^2 < 1\ 520$), indicating high densities of few instability indicators like *Chaetozone sp.* (315 ind/m^2 in S26 and 215 ind/m^2 in S8).

IV. Transitional zone (corresponds to subnormal III of Bellan, 1985). The diversity and evenness indices increase more while more sensitive species are encountered. Higher diversity even close to normal but several instability indicators can be significantly abundant (*Paralacydonia paradoxa, Lumbrineris latreilli, Nematonereis unicornis*). Stations S11, S13, S16 of the inner Saronikos can be classified in this zone. Diversity ranges higher than the previous zone ($4.34 < H < 4.64$), also evenness is fairly high to very high ($0.83 < J < 0.93$), species richness reaches maximal numbers $S = 49$ in Station S13. Only some fairly high densities (1 385 ind/m^2 in S13) indicate the high densities of some instability indicators like *Paralacydonia paradoxa* (135ind/m2 in S13) and *Chaetozone sp.* (190 ind/m^2 in S13). A subdivision among these three stations can be made assigning Station 13 to the internal subzone of Bellan (1985) and Stations S11 and S16 to the external subzone which is closer to normal. This subdivision is justified mainly by the more moderate presence of instability indicators (abundance lower than 100 ind/m^2), and the subsequent higher evenness and relative total density compared to S13.

Figure A2.8 **Pollution zones in Saronikos Gulf based on benthic community studies**



4. Discussion

Table 2 presents some examples of the range of the Shannon diversity index in various regions in disturbed and undisturbed benthic communities in the western Mediterranean and the Atlantic European coasts. It gives the key species and the characterisation of the respective pollution gradients reported from Mediterranean, Atlantic and Pacific areas. Two main conclusions can be derived from this table.

1. The range of the Shannon diversity index should be used as a tool of pollution evaluation, taking into account not only the substrate and depth of the given area but also the regional standards of the case area. For example, the diversity index in undisturbed areas of a mixed sediment infralittoral (22 m) community in the western Mediterranean Ionian Sea reaches the maximum of 4.5 approximately while an analogous community in the eastern Mediterranean (Petalioi Gulf, Aegean Sea, Greece) reaches the maximum value of 6.11 units (Simboura et al., 1998).

2. The key species characterising a pollution gradient may be different when different geographical areas are examined. It is apparent that the dissimilarity in the key species identity is more pronounced among more distant areas, for example among Mediterranean or European Atlantic and Pacific regions (Los Angeles, San Francisco). It is noteworthy that genetic studies have proved that some apparently cosmopolitan key species such as *Capitella capitata* are rather complexes of sibling species with geographical differentiation.

Conclusively, indicators (diversity index level and key species) can be used as effective tools only when a background data set is available pertaining to the specific ecosystem.

Table A2.2

Examples of the range of the Shannon diversity index in various regions in disturbed and undisturbed benthic communities in the western Mediterranean and the European Atlantic Ocean

Region	Depth (m)	Substrate	Indicators/abundant species	H	Zone/community
Golden Horn Izmir (Unsal, 1988)	2–40		*Capitella capitata* *Malacoceros fuliginosus* *Polydora ciliata*	0.11 –2.02	Highly polluted (II)
La Coruna (Lopez-Jamar et al., 1995)	16	Mud	*Thyasira flexuosa* *Chaetozone sp.* *Capitella capitata*	1.23 –3.76 2.75 mean	Highly dredged, hypoxic
La Coruna (Lopez-Jamar et al., 1995)	9	Well-orted fine sand	*Paradoneis armata* *Tellina fabula* *Spio decoratus*	2.70 –4.33 3.42 mean	Relatively clean
Ionian Sea (W. Mediterranean) (Alberteli et al., 1995)	30–100 100–200 > 200	Soft bottom	*Amphipods, echinoid* *Loimia meduda,* *Spiochaetopterus costarum* *Sabellides octocirrata,* *Spiochaetopterus costarum*	4.5 3.6 3.2	Undisturbed
Ionian Sea (Apulian coasts, Italy) (Bedulli et al., 1986)	63 18 49 22 22	Mud Mud Muddy sand Sand Muddy sand	*Nephtys hystricis, Sternaspis scutata* *Corbulla gibba, Notomastus aberans, Tellina distorta* *Modiolula phaseolina* *Apseudes latreilli, Amphipoda, Aponuphis bilineata* *Owenia fusiformis, Lumbrineris latreilli, L.gracilis*	3.9 3.8 4.2 4.4 4.5	Unstable, transitional
From Peres and Bellan, 1972					
San Francisco Los Angeles Marseilles Saltkallefjord Finland			*Capitella, Neanthes succinea* *Capitella capitata* *Capitella capitata, Scolelepis fuliginosa* *Capitella capitata, Scolelepis fuliginosa* *Nereis, macoma, Chironomidae*		Marginal or polluted
Los Angeles Marseilles San Francisco Finland			*Cirriformia luxuriosa, Capitella* *Nereis caudata, Cirriformia tentaculata,* *Schistomeringos rudolphii* *Streblospio benedicti* *Tubifex, Mya*	Intermediate zone of pollution	Semi-healthy II
Los Angeles Marseilles San Francisco Finland Saltkallefjord			*Dorvillea articulata, Capitella present* *Tharyx parvus, Cossura candida, Polychaetes* *Polychaetes and molluscs with wide ecological requirements* *Mya arenaria, Macoma incospicua* *Corophium, Mesidothea* *Amphiura filiformis, A. chiajei*		Semi-healthy I 'healthy-enriched' subnormal zone
Saltkallefjord Finland			*Maldane filiformis and Melesina tenuis community* *Harmothoe, Cardium*		Healthy zone, external zone

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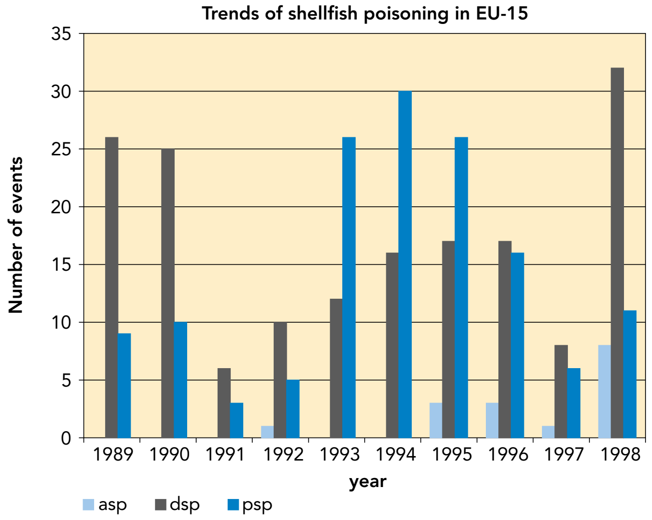
Annex 3

Phytoplankton and phytotoxins indicators

Indicator fact sheet

Figure A3.1 **Changes in ASP, DSP and PSP in EU-15 from 1989 to 1998**

Source: ICES-IOC HAEDAT (harmful algae event database).



Note: For trends in shellfish poisoning concentration expressed as a sum of number of events of ASP, DSP or PSP in each country of EU-15, see Table 1.

Indicator	Policy issue	DPSIR	assessment
Phytoplankton and phytotoxins	Do we see the area where policies are useful to protect human health?	State	😐

Through the numerous phytoplankton species that exist all over the world, several toxic or harmful species have been recorded. Among toxic species, some of them produce toxins directly toxic to marine fauna, and other produce toxins which accumulate in shellfish, fish, etc. They may subsequently be transmitted to humans, and through consumption of contaminated seafood become a serious health threat. Therefore toxins are searched for in shellfish with the aim to protect consumers.

Five human syndromes are presently recognised to be caused by consumption of contaminated seafood:

- amnesic shellfish poisoning (ASP),
- ciguatera fish poisoning (CFP),
- diarrhetic shellfish poisoning (DSP),
- neurotoxic shellfish poisoning (NSP),
- paralytic shellfish poisoning (PSP).

The 15 Member States of the European Union (EU-15) are only concerned with three of them: ASP, DSP and PSP.

😐 Shellfish poisoning events show fluctuation between years without general trend for EU-15.

Target: to reduce the risk of serious seafood poisoning, intensive monitoring of the species composition of the phytoplankton is required in the harvesting areas in connection with bioassays and/or chemical analyses of the seafood products.

Regulation texts: Council Directive 91/492/EEC of 15 July 1991, modified by Council Directive 97/61/EEC of 20 October 1997, fixes the health conditions for the production and the placing on the market of live bivalve molluscs. The main phytotoxins observed in EU-15 are PSP toxins (maximum level 80µg equivalent STX (1) per 100 g of

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(1) STX = Saxitoxin.

Number of events involving ASP, DSP, or PSP toxins in EU-15, by country, 1989–1998 Table A3.1

Year	A	B	D	DK	E	EL	F	FIN	I	IRL	L	NL	P	S	UK	EU-15 ASP	EU-15 DSP	EU-15 PSP
1989			0	0	6		17			0		3	9	0	0	0	26	9
1990			1	0	6		14			0		1	10	1	2	0	25	10
1991			0	0	1		5			0		0	0	0	3	0	6	3
1992			0	0	2		9			0		0	0	1	4	1	10	5
1993			0	0	3		8			0		0	24	1	2	0	12	26
1994			1	1	4		2			1		2	29	0	6	0	16	30
1995			0	0	6		2			2		1	33	1	1	3	17	26
1996			0	1	3		0			2		0	21	0	10	3	17	16
1997			2	0	5		2			1		0	1	2	1	1	8	6
1998			1		12		3			2		0	16	1	16	8	32	11

Source: HAEDAT.

flesh), ASP toxins (maximum level 20µg of domoïc acid per g of flesh), DSP toxins (no positive reaction for biological test). Overviews on events of their presence (irrespective of the level of toxicity) in EU-15 for 1989–98 are included in the ICES countries maps (Maps 1 to 3).

Evaluation: Levels and trends (Figure A3.1) of each shellfish poisoning since 1989 to 1998 for EU-15 show that:

- ASP is episodic with one peak over five events in 1998;
- PSP presents important peaks from 1993 to 1995, over 25 events, then decreases. However the beginning of a new increase appears in 1998;
- DSP was the most regularly observed with a number of events always over five, with three peaks equal or above 25 in 1989, 1990, and 1998.

The results of all toxic events observed are summed up for EU-15 since 1989 in Table A3.1.

To conclude, shellfish poisoning in all countries has increased since 1998, showing the importance of maintaining surveillance measures, especially to protect the consumers of shellfish.

Meta data

Technical information

1. Data sources: ICES-IOC harmful algae event database (HAEDAT).
2. Description of data: ASP, PSP, DSP
   Original name of the data file (with links): HAEDAT (http: // ioc.unesco.org/hab/data33.htm)
   Original measurement units: occurrences of harmful algae events (HAEs)
   Original purpose: public health
3. Geographical coverage: ICES area.
4. Temporal coverage: 1989–98.
5. Methodology and frequency of data collection: monitoring of phytoplankton and phytotoxins.
6. Methodology of data manipulation: sum of numbers of event for each year.

Qualitative information

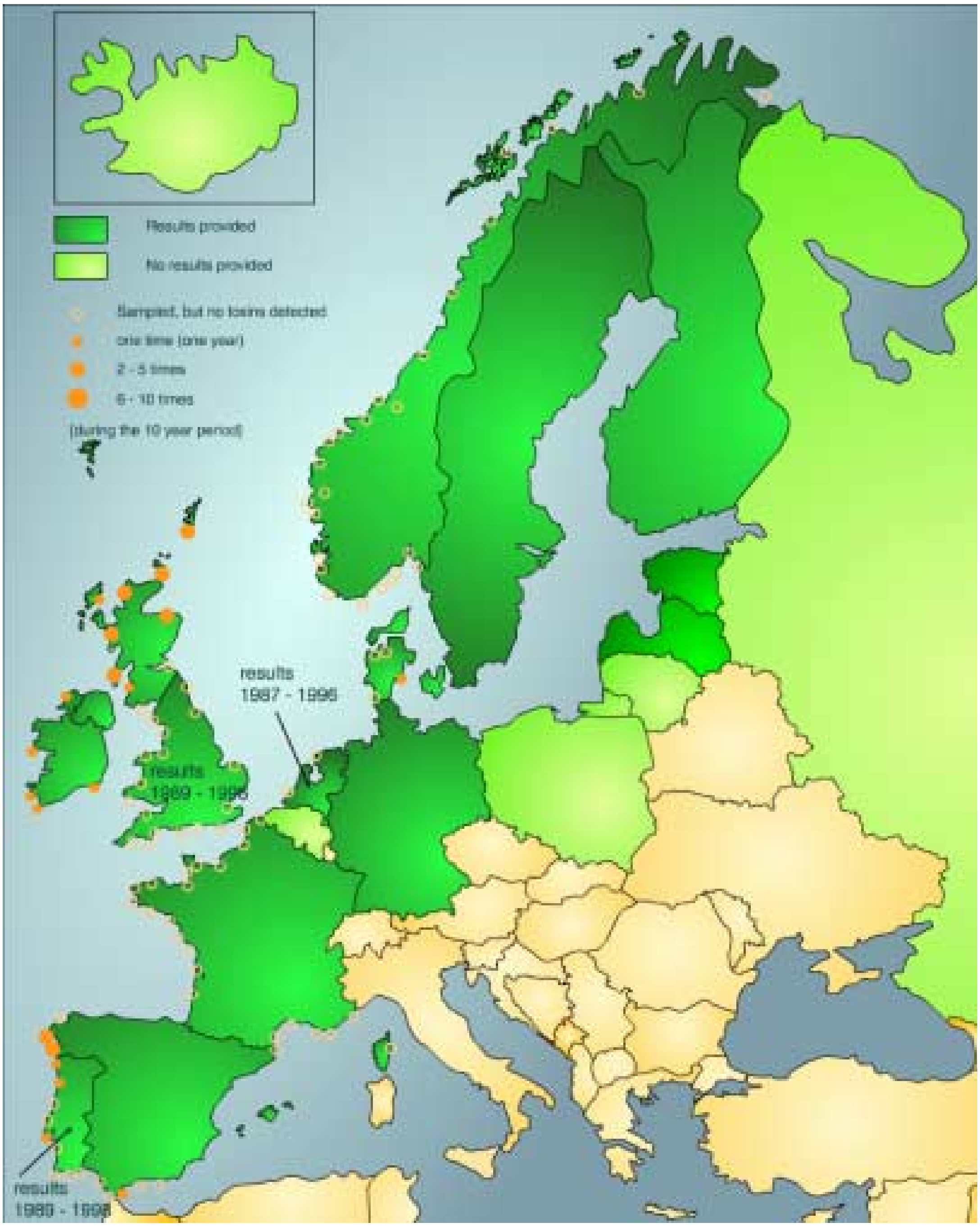
7. Strength and weakness (at data level): available information on individual events varies greatly from event to event or country to country (disclaimer from ICES).
8. Reliability, accuracy, robustness, uncertainty (at data level): to be filled in at assessment.
   Monitoring intensity, number of monitoring stations, number of samplings, stations, etc. also vary greatly and therefore there is not a direct proportionality between recorded events and actual occurrences of, for example, toxicity in a given region. Furthermore, areas with numerous recorded occurrences of HAEs, but with an efficient monitoring and management programme, may present a low risk of intoxications, whereas rare HAEs in other areas may cause severe problems and could present significant health risks. Therefore, these maps should thus be interpreted with caution regarding risk of intoxication by seafood products from the respective areas/regions/countries. IOC and ICES are not liable for any possible misuse of this information.
9. Further work required (for data level and indicator level): decadal data and maps, one year after.

References: HAEDAT (ICES-IOC harmful algae event database).

Signs used for the assessment:

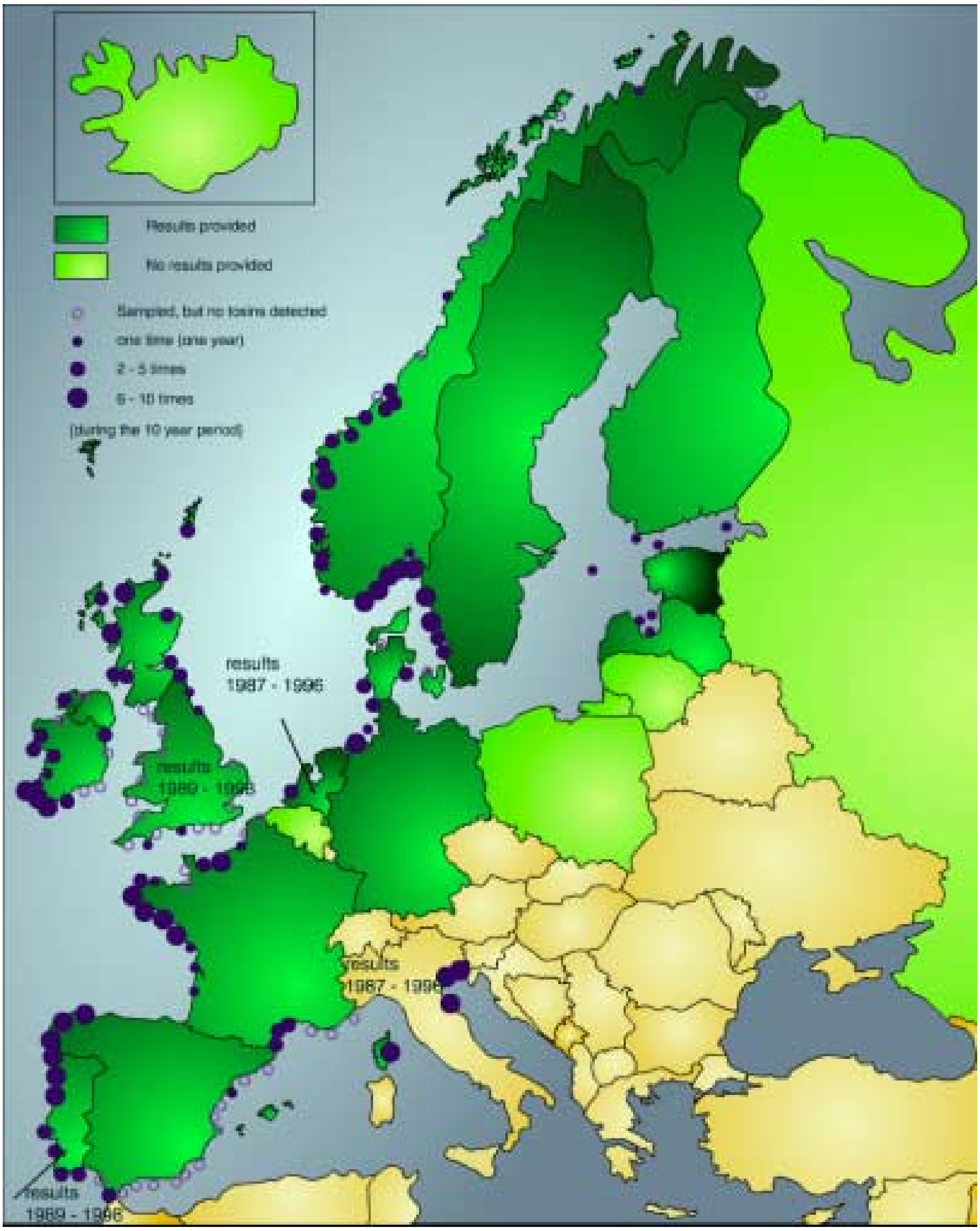
☹ **increase** 😐 **equal** ☺ **decrease**

Map 1 Toxic serial cases for ASP, 1990–99



Toxic serial cases for DSP, 1990–99

Map 2



Map 3 **Toxic serial cases for PSP, 1990–99**