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Eutrophication in the Baltic Sea

An integrated thematic assessment of the effects of nutrient enrichment in the Baltic Sea region

Helsinki Commission

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PREFACE

The environment of the Baltic Sea is unique and fragile. Pollution, fishing, physical modifications, and other human activities exert pressure on a large variety of marine habitats and species. The populations of the countries bordering the Baltic Sea are concerned about its environmental status. Because most of the threats are trans-boundary, solutions must be trans-national. Therefore, the Baltic Sea countries have joined forces in order to safeguard the Baltic Sea environment and to coordinate mitigative efforts. The framework for this work is the 'Convention on the Protection of the Marine Environment of the Baltic Sea Area' – known as the Helsinki Convention. The governing body is the Helsinki Commission, which is responsible for the coordination of activities and day-to-day work.

For more than three decades, the Helsinki Commission (HELCOM) has coordinated monitoring and assessment activities in the Baltic Sea region. In 2005, the Commission adopted a new Monitoring and Assessment Strategy. As a result, focus of the assessment activities shifted towards integrated thematic assessments, which concentrate on specific issues and are more detailed and solution-oriented than the previous, broad Periodic Assessments of the State of the Marine Environment in the Baltic Sea Area published in 1980, 1987, 1990, 1996, 2002 and 2003.

The effects of nutrient enrichment, also known as eutrophication, are perhaps the single greatest threat to the Baltic Sea environment. Understanding of eutrophication becomes clearer when one considers the origin of the word, which has its root in two Greek words: 'eu' which means 'good' or 'well', and 'trope' which mean 'nourishment'. Consequently, a translation would be 'well nourished', but the modern use of the word eutrophication is related to excess loads of nutrients, nutrient enrichment, and adverse effects in aquatic ecosystems. Nutrient enrichment results in an increase in productivity and undesirable changes in ecosystem structure and functioning. Marine systems such as the Baltic Sea can cope with the increases to some extent. When the limits of 'normal' ecosystem structure and functioning are exceeded, however, the ecosystem as a whole is confronted with the problem of eutrophication.

This report describes and documents the degree and effects of nutrient enrichment and eutrophication in the Baltic Sea including the Kattegat/ Belt Sea area. The objectives of this eutrophication assessment are:

- To define the issue, by answering the questions: 'What is nutrient enrichment and eutrophication?' and 'How are nutrient loads, nutrient concentrations, biological quality elements and other effects interlinked?'
- To document the eutrophication status of the Baltic Sea by focusing on the following chemical and biological quality elements: nutrients, phytoplankton, water transparency, submerged aquatic vegetation, oxygen concentrations, and benthic fauna.
- To document the causes of eutrophication by describing nutrient loads (waterborne and airborne) to the Baltic Sea including their sources.
- To discuss solutions to the eutrophication problems in the Baltic Sea, e.g. by assessing existing national and Baltic-wide strategies, actions and measures to combat eutrophication, and by outlining supplementary measures required to reduce eutrophication to acceptable levels also taking into account future challenges related to changing environmental conditions and human pressures.

This report is directly linked to the HELCOM Baltic Sea Action Plan, which identified eutrophication as one of the four main issues to address in order to improve the environmental health of the Baltic Sea. The Action Plan sets a strategic goal related to eutrophication, namely, 'Baltic Sea unaffected by eutrophication', and identified a set of Ecological Objectives which corresponds to good ecological/environmental status. This thematic assessment addresses each of the Ecological Objectives for eutrophication and enables an evaluation of progress towards the Ecological Objectives.

In the Baltic Sea Action Plan, the Contracting Parties acknowledged that a harmonized approach to assessing the eutrophication status of the Baltic Sea is required. Therefore, the Contracting Parties agreed to further develop a common HELCOM assessment tool for use in a Baltic-wide thematic assessment of eutrophication in coastal as well as open sea waters.

Ecological objectives related to eutrophication were adopted in the HELCOM Baltic Sea Action Plan. They are: concentrations of nutrients close to natural levels, clear water, natural level of algal blooms, natural distribution and occurrence of plants and animals, and natural oxygen levels.

In some coastal areas, the classification presented in the Baltic Sea-wide eutrophication assessment cannot be directly compared to the results of national assessments and the Baltic Sea intercalibration exercise *sensu* the Water Framework Directive owing to differences in spatial and temporal scaling, as well as the use of parameters that are considered supporting in WFD.

This thematic eutrophication assessment is aimed at decision-makers, managers, scientists, educators and others interested in the health status of the Baltic Sea; it includes a glossary in order to support readers without a professional background in marine ecology or oceanography. The assessment is supplemented by a technical Background Report as well as an Executive Summary which are available via http://www.helcom.fi.

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1 INTRODUCTION

The world's oceans and coastal waters hold pure water and a variety of chemical and biological components. These include organisms, dissolved and dispersed gases, and minerals: the latter includes nutrients originating from geological weathering as well as from human activities. However, nutrient loads to coastal waters have significantly increased in recent decades, mostly because of population growth and changes in nutrient utilization, transfer and management in catchment areas.

The effects of increased nutrient loads and the resulting nutrient enrichment have been documented on a global scale. However, effects are most pronounced in regional seas which have a combination of a high population density in the catchment and physiographic characteristics predisposing the sea to nutrient enrichment.

In Europe, nearly all regional seas have faced increased loads and nutrient enrichment and have witnessed the undesirable effects of eutrophication. One of the most prominent and direct effects is an increase in phytoplankton productivity and biomass, often 'illustrated' as chlorophyll-*a* concentratios.

The Baltic Sea has both similarities and differences compared to other European seas with regard to chlorophyll-*a* concentrations (**Fig. 1.1**). In the northernmost part of the Baltic Sea, chlorophyll-*a* concentrations are lowest, as is the population density, whereas the eastern, western and southern parts of the Baltic Sea are characterized by relatively high chlorophyll-*a* concentrations resulting from large nutrient supplies and nutrient enrichment in these areas.

A high chlorophyll-*a* concentration is considered to represent a eutrophication signal and to be an indication of areas affected by eutrophication. Hence, it would be tempting to conclude that the whole Baltic Sea is affected by eutrophication. However, things are not that straightforward. Because marine systems are dynamic in nature, there are differences in their sensitivity to nutrient enrichment, and eutrophication is manifested in a variety of ways. This is why an integrated assessment of eutrophication processes and status in the Baltic Sea is needed.

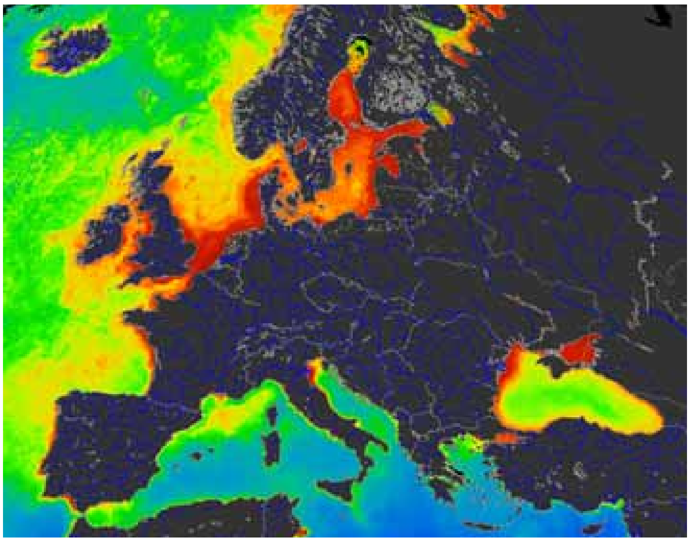


Figure 1.1 Satellite image of chlorophyll-*a* concentrations in European seas. Red indicates high concentrations while yellow, green and blue indicate successively lower concentrations. Source: JRC, http://marine.jrc.ec.europa.eu/.

Management actions are most efficient if based on well-documented facts and figures, as well as jointly agreed principles. So before jumping to any premature conclusions, we need to sort out three things. Firstly, to agree on what we are talking about, i.e. what we mean by nutrient enrichment and eutrophication. Secondly, to document what are the sources of nutrient enrichment and eutrophication, and what their root causes are. Thirdly, to discuss and provide guidance on how the countries bordering the Baltic Sea could improve management of the undesirable effects of eutrophication. This is the aim of the Baltic Sea-wide assessment of eutrophication status.

1.1 What is nutrient enrichment and eutrophication?

Briefly, eutrophication means 'well nourished', but for more than three decades it has been acknowledged that excessive amounts of nutrients, nitrogen (N), phosphorus (P), and sometimes organic matter (represented by carbon, C), can result in a series of undesirable effects.

The major effects of eutrophication include changes in the structure and functioning of the entire marine ecosystem and a reduction in ecosystem stability. The first response to increased nutrient loads is a corresponding increase in nutrient concentrations which tends to take place despite naturally occurring variations in runoff and precipitation.

Another effect is a change in the ratio between dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) in the water. The optimal DIN:DIP ratio (N:P ratio) for phytoplankton growth is 16 moles to 1 mole; this is the so-called Redfield ratio. A significantly lower N:P ratio indicates potential nitrogen limitation, while a higher N:P ratio implies potential phosphorus limitation of phytoplankton primary production. Deviations from the Redfield ratio may affect primary production, the concentration and quality of phytoplankton biomass, species composition, and consequently food-web dynamics.

1.2 Why is the Baltic Sea sensitive to eutrophication?

The Baltic Sea is the only inland sea wholly in Europe and is one of the largest brackish-water basins in the world. It is divided into several sub-regions and includes a transition zone to the North Sea (the Belt Sea and Kattegat area) consisting of sub-basins separated by sills. The major sub-basins of the Baltic Sea (**Fig. 1.2, Table 1.1**) are: (1) the Baltic Proper (Arkona Basin, Bornholm Basin, western and eastern Gotland Basin, Gdansk Deep, Northern Baltic Proper), (2) the Gulf of Bothnia, comprising the Bothnian Sea and the Bothnian Bay, (3) the Gulf of Finland, (4) the Gulf of Riga, and (5) the Danish Straits, including the Belt Sea, and (6) the Kattegat area. The Baltic Sea has an average depth of 52 m, with a volume of 21,700 km^3 and a surface area of 415,200 km^2. The different sub-basins or sub-areas of the Baltic Sea differ considerably in their characteristics from north to south and from east to west, for example, in terms of ice cover, temperature, salinity, and residence time of the water.

The sub-basins differ not only in size, volume, and depth, but also in the salinity of the water, which is very important for the biota. In the southwestern parts of the Baltic Sea (Kattegat), surface layer salinity normally is 20–25; in the central Baltic Proper, it is 6–8; in areas such as the northern and eastern extremities of the Bothnian Bay and the Gulf of Finland, respectively, salinity may drop to below 1.

The combination of a large catchment area with associated human activities and a small body of water with limited exchange with the Skagerrak and the North Sea makes the Baltic Sea very sensitive to nutrient enrichment and eutrophication. The catchment area of the Baltic Sea is more than 1,700,000 km^2, with a population of approximately 85 million inhabitants. The population density varies from less than 1 person per km^2 in the northern and northeastern parts of the catchment area to more than 100 persons per km^2 in the southern and southwestern parts. The land-use structure follows the same pattern as the population density, with a high proportion of arable land in the eastern, southern, and western parts, and predominantly forest, wetlands and barren mountains in the north.

The combination of a high population density, a well-developed agricultural sector, and other human activities, such as emissions from energy production and transport, has resulted in large loads of nutrients, mainly compounds of nitrogen and phosphorus, entering the Baltic Sea. This has resulted in the problems and challenges of eutrophication in the Baltic Sea.

As already mentioned, the limited water exchange with the Skagerrak and North Sea, and the resulting long residence time of water are the main reasons for the sensitivity of the Baltic Sea for eutrophication. High nutrient loads in combination with long residence times means that nutrients discharged to the sea will remain for a long time, sometimes for decades, before being flushed out of the Baltic Sea into the Skagerrak surface waters.

A physical feature which markedly increases the vulnerability of the Baltic Sea is the vertical stratification of the water masses. Stratification is caused by differences in salinity of the bottom and top layers of the water column and seasonally by differences in water temperature. The most important effect of stratification in terms of eutrophication is that it hinders or prevents ventilation and oxygenation of the bottom waters and sediments by vertical mixing of water, a situation that often leads to oxygen depletion. Hypoxia and anoxia have an effect on nutrient transformations, such as nitrification and denitrification processes, as well as the capacity of

Figure 1.2 Map of the Baltic Sea with catchment areas. The Baltic Proper embraces the northern Baltic Proper, Western Gotland Basin, Eastern Gotland Basin, Gulf of Gdansk, Bornholm Basin and Arkona Basin. The Gulf of Bothnia comprises the Bothnian Sea, the Quark and the Bothnian Bay. The Danish Straits include the Sound, Great Belt and Little Belt.

the sediments to bind phosphorus. In the absence of oxygen, reduced sediments release significant quantities of phosphorus to the overlaying water.

Another feature which makes the Baltic Sea sensitive is the low salinity, and the intermediate nature of the Baltic Sea, being neither a true oceanic nor a freshwater environment. The number of naturally occurring species in the different parts of the Baltic Sea is influenced by the salinity gradients from the saline North Sea to the brackish northern and eastern parts (**Fig. 1.3**). The maximum number of species is found at salinities over 30. When the salinity decreases, so does the number of marine species. The minimum number of species is found at salinities in the range of 8–10, which is the typical salinity in many parts of the Baltic Sea. At salinities below 8, the number of species increases because some species which typically live in freshwater can cope with these salinities.

Table 1.1 **Physical characteristics of Baltic Sea basins. Based on Andersen & Pawlak (2006) and Meier (2007).**

		Area	Max. depth	Ave. depth	Volume	River water, mean age [1]	Marine water, mean age [2]
		km^2	m	m	km^3	Years	Years
1	Gulf of Bothnia	115,516	230	60	6,389	2–28	34–42
2	Gulf of Finland	29,600	123	3	1,100	2–24	24–32
3	Gulf of Riga	16,300	> 60	26	424	16–24	30–32
4	Baltic Proper	211,069	459	62	13,045	28–30	10–28
5	Danish Straits	42,408	109	19	802	28–30	8–10
	Total Baltic Sea	414,893	459	52	21,760	-	-

1) Mean age of riverine water in the surface layer of each basin. The residence of riverine water is some years longer if mixed to deeper layers.

2) Mean age of marine water (from Danish Straits) in the bottom layers of the basins. The residence time of marine water in that layer is shorter than in the surface layer.

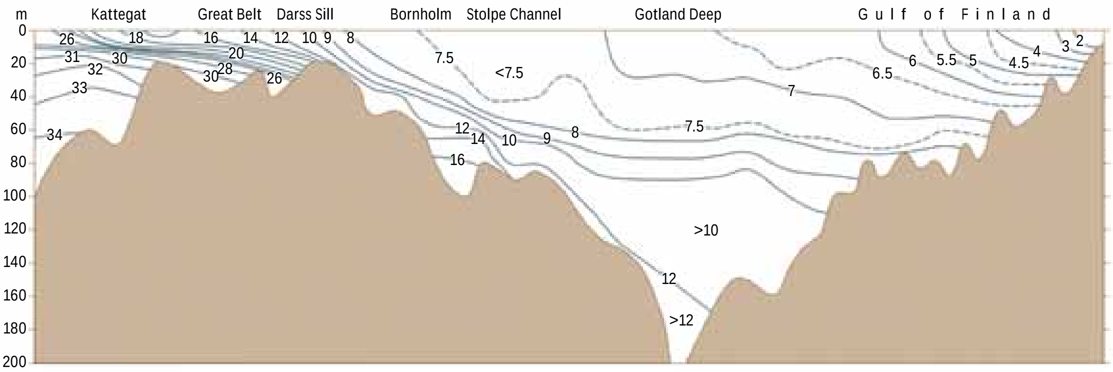
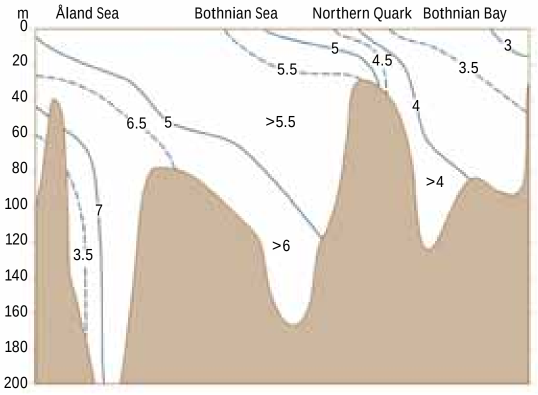


Figure 1.3 **Transect of the Baltic Sea from the Kattegat through the Danish Straits and the Baltic Proper into the Gulf of Finland (upper panel). A transect of the Gulf of Bothnia from Åland Sea to the Bothnian Bay is shown in the lower panel. The isolines show salinity levels. Graph modified from Leppäranta & Myrberg (2008).**



1.3 How are loads, nutrient concentrations and eutrophication effects coupled?

The manifestations of a large-scale eutrophication problem are well known in most parts of the Baltic Sea. These include murky water owing to blooms of planktonic algae, mats of macroalgae at shores, reduced distribution of benthic habitats such as eelgrass meadows, and oxygen depletion resulting in the death of benthic animals and fish.

It is essential to keep in mind that the undesirable effects of eutrophication are related to human activities that give rise to increased nutrient loads, which lead to nutrient enrichment and elevated nutrient concentrations in the sea. It is also essential to understand that once the nutrients are in sea areas with certain characteristics, in amounts that increase concentrations, they are likely to stimulate the growth of plants and algae living in those ecosystems.

In coastal waters, nutrient enrichment will generally cause an increase in phytoplankton primary production and the growth of short-lived macroalgae. An increase in phytoplankton biomass will result in a decrease in light penetration through the water column. Decreased light penetration, which is often measured as a decrease in water transparency (visibility) by a 'Secchi depth' instrument, can ultimately reduce the colonization depth of macroalgae and seagrasses.

Primary production in offshore water is most often limited by the availability of light and nutrients. However, dissolved inorganic carbon is also a prerequisite, whilst temperature affects rates of primary production. Nutrient enrichment will generally cause an increase in phytoplankton primary production, resulting in an increase in phytoplankton biomass and ultimately an increase in sedimentation of organic matter to the seafloor. The general responses of pelagic ecosystems to nutrient enrichment can, in principle, be a gradual change towards: (1) increased planktonic primary production compared to benthic production, (2) a dominance of microbial food webs over the 'classic' planktonic food chain from small to large organisms, (3) a dominance of non-siliceous phytoplankton species over diatom species, and (4) a dominance of gelatinous zooplankton (jellyfish) over crustacean zooplankton, (5) increased sedimentation of organic matter to the seafloor, (6) near-seafloor oxygen depletion caused by oxygen consumption by the degrading organic matter, ultimately resulting in hypoxia or anoxia, and (7) loss of higher life forms, including fish and bottom invertebrates owing to poor oxygen conditions. Such changes may also be driven by loads of organic matter via riverine or direct discharges.

Currently, large parts of the Baltic Sea are in a state of so-called repressed recovery, sometimes referred to as a vicious cycle, because of the interconnected processes involving nitrogen, phosphorus and oxygen (cf. Vahtera et al. 2007 and **Fig. 1.4**). Widespread hypoxia facilitates the release of phosphorus from the sediment and fuels blooms of nitrogen (N_2) fixing blue-green algae that tend to counteract reductions in external P and N loads.

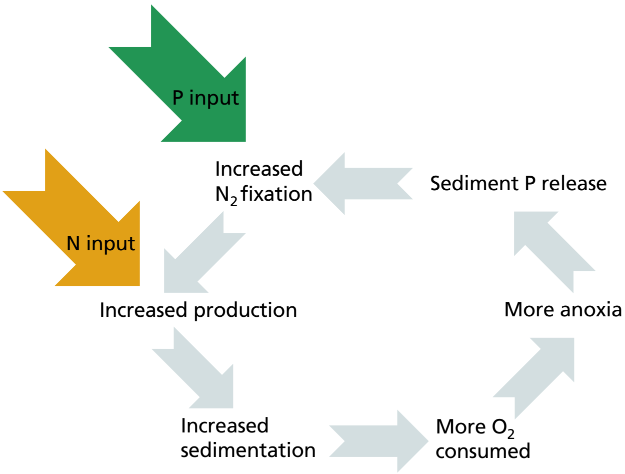


Figure 1.4 A simple conceptual model of the positive feedbacks between nitrogen and phosphorus in the Baltic Sea. Based on Elmgren (2006).

Phytoplankton production is, in general, limited by the least-abundant growth factor, e.g. light, carbon or nutrients. With respect to estuarine and marine ecosystems, the nutrients in question are nitrogen and/or phosphorus. Sometimes other nutrients, such as silica (SiO_2) or trace elements such as iron (Fe) can become limiting. It has been debated whether nitrogen or phosphorus is the most limiting nutrient in the Baltic Sea. There is a general consensus that both nutrients matter but their importance varies between basins and seasons. It is widely acknowledged that improvement of the eutrophication status of the Baltic

Sea calls for further reductions in nitrogen load, as primary production in the open basins is mainly nitrogen limited, except for in the Bothnian Bay and Bothnian Sea. Nitrogen limitation is particularly true for the spring period, with the spring phytoplankton bloom forming the biomass peak of the year. Undesirably large summer blooms of nitrogen-fixing cyanobacteria are driven by excess phosphorus, along with high temperatures. They also contribute tremendous quantities of new nitrogen to the pelagic ecosystem making it doubly important to reduce phosphorus loads along with those of nitrogen.

1.4 Assessment principles and data sources

The elaboration of internationally harmonized assessments of the Baltic Sea has been central to HELCOM's work since the beginning of the 1980s. The HELCOM Baltic Sea Action Plan, adopted in late 2007, addresses eutrophication and sets a vision, strategic goals and a suite of ecological objectives which correspond to good ecological/environmental status. HELCOM monitoring and assessment activities aim to indicate how the vision, and the goals and ecological

objectives for the Baltic Sea marine environment are being met. Assessments also couple the quality of the environment to management actions. This strategy aims to promote an operational assessment system with annual indicator fact sheets and regular thematic assessments leading to holistic assessments covering all aspects of the status of and pressures on the marine environment of the entire Baltic Sea.

The main objective of the HELCOM assessment products is to provide policy-relevant information for targeted users at national and Baltic-wide level, as well as to provide input to pan-European and global fora (EU, UNEP, IMO). This is necessary in order to make sound decisions to restore the Baltic Sea ecosystem, to achieve and maintain good ecological status, and to support the implementation of the HELCOM objectives and actions. An essential objective is to raise general public awareness of the Baltic Sea and of HELCOM actions.

This HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea consists of a technical/scientific (science for management) section and a policy implication section. It is to a large extent based on data produced within the HELCOM COMBINE monitoring programme and it should therefore be seen as a link in a continuous chain towards holistic assessments where research, monitoring, and modelling results published in scientific reports play an important role in explaining and linking pressures, state, and impacts and providing guidance for future policy or management responses.

When documenting the eutrophication status, the assessment puts focus on nutrient loads and concentrations, the status of biological quality elements (phytoplankton, submerged aquatic vegetation and benthic invertebrates), and oxygen conditions. For nutrients, biological quality elements and oxygen conditions, the assessment compares reference conditions with the actual status in the period 2001–2006 and considers temporal trends. In addition, the assessment produces a final and integrated classification of eutrophication status, on the basis of which general conclusions and recommendations are made.

The final classification of eutrophication status in different parts of the Baltic Sea has been made by application of the HELCOM Eutrophication Assessment Tool (HEAT). A total of 189 ‘areas’ (a mix of stations, sites or basins) have been classified as either ‘areas affected by eutrophication’ (having a moderate, poor or bad status) or ‘areas not affected by eutrophication’ (having a high or good status). HEAT is indicator-based and uses the ‘one out – all out principle’ *sensu* the Water Framework Directive, which means that the overall classification of an assessed area is based on the most sensitive quality element. HEAT also estimates a so-called interim ‘confidence’ of the final classification results in order to assess the reliability of the final classification. As a precautionary note, it should be emphasized that the integrated assessment tool HEAT, used for classifying ‘areas affected by eutrophication’ or ‘areas not affected by eutrophication’ makes use of synoptic information in regard to reference conditions, acceptable deviation from reference conditions, and actual environmental status, the latter for the period 2001–2006.

In some coastal areas, the classification presented in the Baltic Sea-wide eutrophication assessment cannot be directly compared to the results of national assessments and the Baltic Sea intercalibration exercise *sensu* the Water Framework Directive owing to differences in spatial and temporal scaling, as well as the use of parameters that are considered supporting in WFD.

It should be noted that this HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea is based on jointly agreed methods and assessment principles. It is scientifically based and the majority of data sets used originate from HELCOM COMBINE or HELCOM PLC-5 as well as research activities. Hence, most data are quality assured and controlled in accordance with HECOM COMBINE guidelines. Furthermore, the indicators used in regard to phytoplankton, submerged aquatic vegetation, benthic invertebrate fauna as well as physico-chemical features and loads have been reported in various national, regional or European assessment reports.

2 WHAT ARE THE EUTROPHICATION SIGNALS?

Eutrophication is a major problem in the Baltic Sea and the Baltic Sea states share an overall goal of having a Baltic Sea unaffected by eutrophication. The countries bordering the Baltic Sea have interpreted and translated this overall goal into a common Baltic Sea-wide vision for good environmental status in the Baltic Sea. With regard to eutrophication, the vision of good environmental status has been divided into the following suite of ecological objectives: 'Concentrations of nutrients close to natural levels', 'Clear water', 'Natural level of algal blooms', 'Natural distribution and occurrence of plants and animals', and 'Natural oxygen levels'.

In order to make the ecological objectives operational, indicators with initial target values have been agreed upon that reflect a good ecological and environmental status of the Baltic marine environment. Thus, the target values, when achieved, are intended to represent good ecological or environmental status.

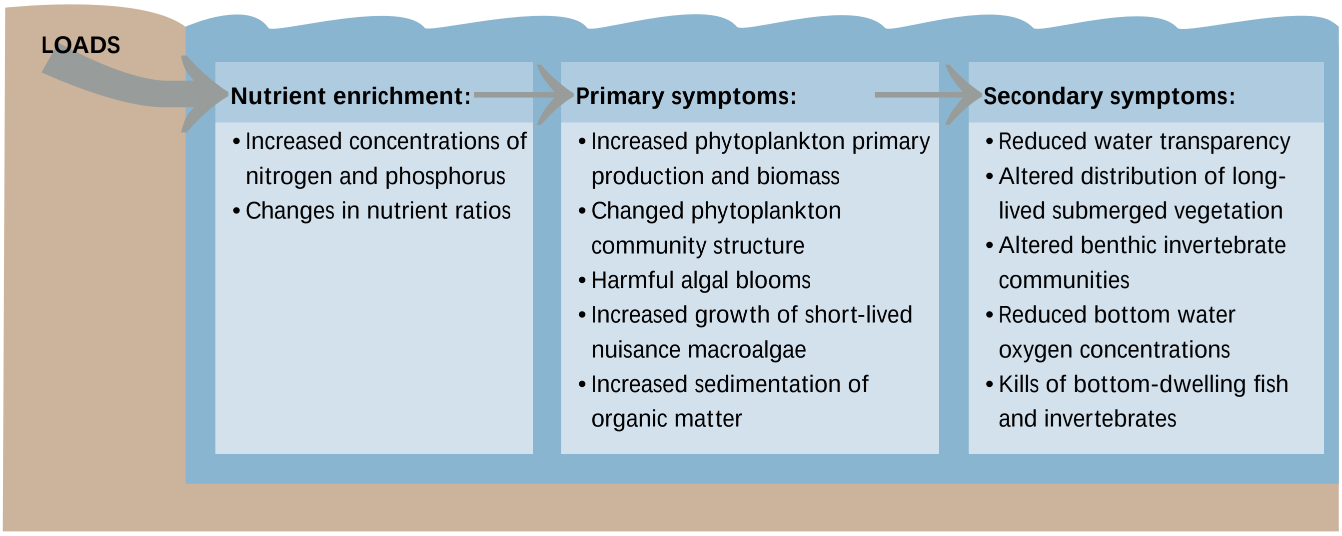
It has been agreed that the ecological objectives for eutrophication will be measured by the following indicators:

- Winter surface concentrations of nutrients, reflecting the ecological objective 'Concentrations of nutrients close to natural levels'.
- Chlorophyll-*a* concentrations, reflecting the ecological objective 'Natural level of algal blooms'.
- Secchi depth, reflecting the ecological objective 'Clear water'.
- Depth range of submerged aquatic vegetation, reflecting the ecological objective 'Natural distribution and occurrence of plants and animals'.
- Abundance and structure of benthic invertebrate communities, reflecting the ecological objective 'Natural distribution and occurrence of plants and animals'.
- Area and length of seasonal oxygen depletion, reflecting the ecological objective 'Natural oxygen levels'.

2.1 Eutrophication signals in focus

The assessment of eutrophication signals focuses on three key issues: (1) status, (2) status expressed as Ecological Quality Ratio, and (3) temporal trends. The difference between an assessment of signal and status versus an assessment of temporal trends is simply that the first focuses on the period 2001–2006, while the second focuses on much longer time spans ranging from several decades up to a century. This HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea focuses in particular on the following quality elements: nutrients, phytoplankton, water transparency, submerged aquatic vegetation, oxygen concentrations, and benthic invertebrate communities. For each quality element, a suite of indicators (sometimes referred to as parameters) is used because they represent a considerable number of the eutrophication signals typical in large parts of the Baltic Sea, cf. **Fig. 2.1**.

The Ecological Quality Ratio (EQR value) has its roots in the Water Framework Directive, which



Nutrient enrichment:

- Increased concentrations of nitrogen and phosphorus
- Changes in nutrient ratios

Primary symptoms:

- Increased phytoplankton primary production and biomass
- Changed phytoplankton community structure
- Harmful algal blooms
- Increased growth of short-lived nuisance macroalgae
- Increased sedimentation of organic matter

Secondary symptoms:

- Reduced water transparency
- Altered distribution of long-lived submerged vegetation
- Altered benthic invertebrate communities
- Reduced bottom water oxygen concentrations
- Kills of bottom-dwelling fish and invertebrates

Figure 2.1 A simple conceptual model of eutrophication symptoms in the Baltic Sea. Based on Cloern (2001).

has the overall goal of achieving or maintaining the good ecological status of surface waters by 2015, a goal almost identical to the goals of the Baltic Sea Action Plan.

A basic step to check whether these goals have been achieved is to conduct an appropriate assessment of the environmental status of waters. Therefore, the values of the biological quality elements must be taken into account when assigning any of the ecological status or ecological potential classes to water bodies. Assessment results are published as Ecological Quality Ratios (EQRs), which represent the relationship between the values of the biological parameters observed for a given body of surface water and the values for these parameters in the reference conditions applicable to that body. The EQR ratio is expressed as a numerical value between one (best) and zero (worst), cf. **Fig. 2.2**.

According to the Water Framework Directive, each EU Member State must divide the ecological quality ratio scale for their monitoring system for each surface water category into five classes ranging from high to bad ecological status by assigning a numerical value to each of the boundaries between the classes.

In practice, high EQR values, close to one, indicate a status with no, minor or slight deviation from reference conditions and hence an acceptable status corresponding to 'areas unaffected by eutrophication'. Low EQR values indicate moderate, major or strong deviations from reference conditions and an unacceptable status corresponding to 'areas affected by eutrophication' having a moderate, poor or bad ecological status.

2.1.1 Nutrients

Nutrients such as nitrate and phosphate are needed for the biomass production of phytoplankton and micro- and macrophytes (e.g. seagrasses, algae). The elements N and P are fixed first in particulate biomass but can be released during transfer in the food chains as dissolved organic compounds (dissolved organic nitrogen (DON), e.g. urea, amino acids, dissolved organic phosphorus (DOP)), ultimately providing nutrient sources for bacteria.

During phytoplankton blooms, inorganic nutrients in surface layers may be almost completely consumed, leading to nutrient limitation. This results in a large seasonal variability of nutrient concentrations. For this reason DIN and DIP are usually measured and assessed during winter, when biological activity is lowest. Total nitrogen (TN) and total phosphorus (TP), which include all forms of N and P compounds, are more robust parameters and can be assessed throughout the year.

Anthropogenic nutrient loads are the origin of eutrophication processes in surface waters, including coastal and offshore areas. Increased biomass production is the first effect, often associated with species shifts, and is usually followed by the

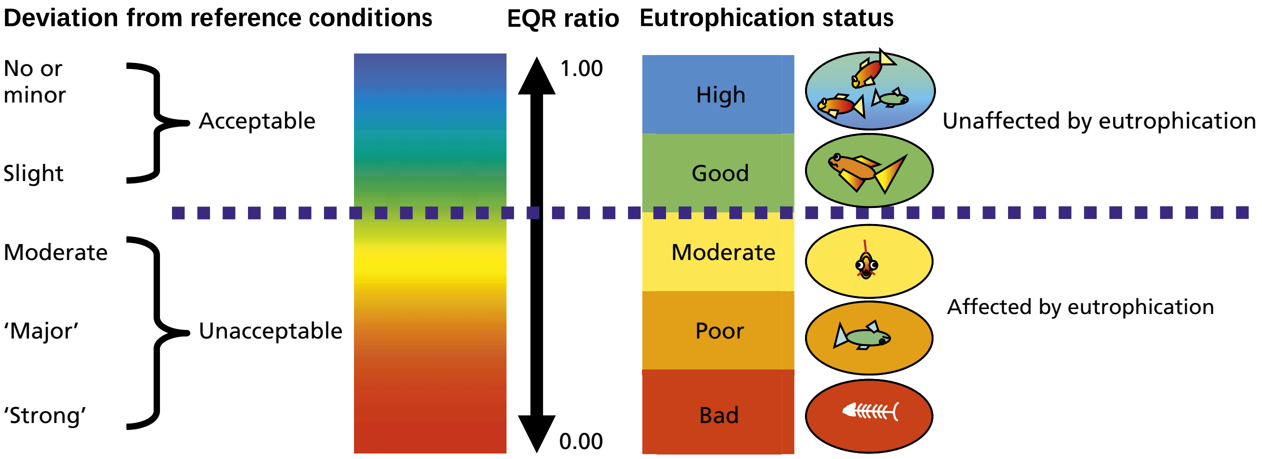


Figure 2.2 Overview of the EQR concept and its use for classifying waters bodies in areas unaffected by eutrophication and areas affected by eutrophication. Based on Anon. (2000) and Anon. (2005). Fish by courtesy of Peter Pollard, SEPA.

accumulation of organic material which results in oxygen depletion in the bottom water of stratified areas after sedimentation. Light limitation is caused by high phytoplankton biomass, reducing the extension of benthic macrophytes and affecting their species composition as well.

A comprehensive assessment of nutrient status and trends in the Baltic Sea is presented in **Chapter 2.2**.

2.1.2 Phytoplankton and water transparency

Planktonic algae, or phytoplankton, are minute unicellular organisms at the base of pelagic food webs in aquatic ecosystems. With generation times ranging from hours to a few days, phytoplankton can respond rapidly to changes in nutrient concentrations and other anthropogenic perturbations.

Phytoplankton reflect planktonic ecosystem productivity and hence are a crucial component in marine and coastal monitoring programmes around the Baltic Sea. Phytoplankton can be monitored in terms of: (1) primary production, (2) biomass (chlorophyll-*a* concentration or carbon biomass), (3) species composition, and (4) bloom frequency, intensity, and spatial and temporal extent.

Algal blooms are a natural phenomenon. Phytoplankton blooms in spring and summer are periods of naturally high-production supplying energy to the ecosystem. Excessive blooms and especially blooms of harmful algae, such as cyanobacteria, are a major problem in the Baltic Sea. Although the sediment record of cyanobacteria dates back 7,000 years, eutrophication increases the intensity and frequency of these blooms. During summer, cyanobacteria form blooms in most parts (both open and coastal) of the Gulf of Finland, the Gulf of Riga, the Baltic Proper, and in the southwestern parts of the Baltic Sea. Such blooms greatly lower the aesthetic and recreational value of the marine environment but they are also potentially toxic to animals as well as humans. Coping with these particular eutrophication effects is one of the most important and, at the same time, most difficult tasks.

Another important effect is that the increased biomass of phytoplankton in water results in increased turbidity and reduced light penetration through the water column to the sea floor. Decreased light hampers the growth of higher plants and macroalgae such as eelgrass and bladderwrack. Murky waters are also unattractive.

At the seafloor, decomposition of sedimented phytoplankton consumes oxygen contributing to the formation of anoxic bottom areas (see **Chapter 2.6**).

A comprehensive assessment of phytoplankton status and trends in the Baltic Sea is presented in **Chapter 2.3**, with a parallel assessment of water transparency in **Chapter 2.4**.

2.1.3 Submerged aquatic vegetation

Extensive seagrass meadows and perennial macroalgal communities harbour the highest biodiversity in coastal, shallow-water ecosystems.

Eutrophication has complex effects on submerged aquatic vegetation (SAV): (1) reduced light penetration through the water column caused by increased pelagic production limits the depth penetration of SAV species, (2) increased sedimentation can prevent the settlement of new specimens on the seafloor and reduces the amount of suitable substrate to be colonized by perennial species on all types of bottom substrates, (3) the excess of nutrients during the whole vegetation period often favours opportunistic species with a short life cycle and rapid development over the perennial species with lower productivity, thus causing a shift in community composition.

The massive development of opportunistic species is the main cause of shifts in the community structure and habitat quality, both in hard- and soft-bottom areas. In worst cases, the biomass accumulated in short-lived filamentous species can serve as an additional local source of nutrients in the eutrophication process while huge masses of detached algae can be transported by currents to locations far from their original growth area.

Chemical and biological degradation of this algal biomass can cause local oxygen depletion that severely affects benthic organisms and communities.

The predictable reactions of SAV communities to increased eutrophication allow several SAV community characteristics to be used as indicators for eutrophication status or as a means of identifying the phase of the eutrophication process.

A comprehensive assessment of the status and trends of submerged aquatic vegetation in the Baltic Sea is found in **Chapter 2.5**.

2.1.4 Oxygen

Oxygen depletion is a common effect of eutrophication in the bottom waters of coastal marine ecosystems and is becoming increasingly prevalent worldwide (HELCOM, 2002). It is caused by the consumption of oxygen by the microbial processes responsible for the degradation of organic matter accumulating at the sea floor. Oxygen depletion may result in hypoxia (literally ‘low oxygen’) or even anoxia (absence of oxygen). These events may be (1) episodic, (2) annually occurring in summer/autumn (most common), or (3) persistent (typical of the deep basins of the Baltic Sea).

In terms of the biological response to hypoxia, the level at which low oxygen concentrations become lethal is species dependent. Fish and crustacea have greater requirements for oxygen and they react very quickly to a shortage. Other species (such as polychaetes and mussels) can tolerate low dissolved oxygen concentrations for longer periods. The benthic responses to hypoxia include a shift from communities of large, slow-growing and slowly reproducing species to communities of small, rapidly reproducing organisms. Anoxic conditions result in the formation of hydrogen sulphide (H_2S), which is lethal to higher organisms.

Oxygen depletion has a clear impact on biogeochemical cycles. Anoxic periods cause the release of phosphorus from sediment. Dissolved inorganic phosphorus (DIP) is significantly negatively correlated with oxygen conditions. The concentration of DIP can vary greatly from year to year depending on the release of

phosphorus from sediments under anoxia (Matthäus et al. 2008). Ammonium is also enriched under hypoxic conditions. The DIP and ammonium from the bottom waters can be mixed into the upper water column and enhance algal blooms. Thus, hypoxia results in large changes in the biogeochemical cycle, which may enhance eutrophication.

Data on long-term trends in Baltic Sea coastal waters have shown that oxygen conditions can be predicted using the values of total nitrogen, temperature, and water exchange. In the Danish Straits, loads of phosphorus have been reduced by 90% and nitrogen by 30%, but periodic hypoxia still occurs (Ærtebjerg et al. 2003). Large-scale hypoxic events were first observed in these areas during 1981 (HELCOM 1990). A change occurred around 1985, with an alteration in benthic communities and the sediment buffering capacity which influenced the remineralization of organic matter. This involved a regime shift to smaller benthic organisms and it does not appear to be reversible.

A comprehensive assessment of oxygen status and trends in the Baltic Sea can be found in **Chapter 2.6**.

2.1.5 Benthic invertebrate communities

The composition of animal communities living on the sea floor of the Baltic Sea reflects the conditions of the environment. In the eutrophication process, broad-scale changes in the composition of the communities – usually involving reduced biodiversity – accompany the increasing organic enrichment of the sediments. At advanced stages of eutrophication, oxygen depletion becomes common.

In many areas of the Baltic, the seafloor animals are exposed to widespread oxygen depletion. These animals have different sensitivities to oxygen depletion; some species can cope for only a short time (from hours to days), while others are able to tolerate oxygen depletion for a longer period (from weeks to more than a month). However, if the oxygen concentration drops below zero and hydrogen sulphide (H_2S) is released, all macrofaunal organisms are eventually killed.

When the oxygen concentration decreases, mobile benthic invertebrates living in the sediment often move to the sediment surface. It is well known that oxygen depletion can lead to increased catches of crustaceans and fish while these animals attempt to escape low oxygen conditions.

There are no benthic invertebrates in areas with permanent oxygen depletion, e.g. in the deep parts of the Baltic Proper. In areas with periodic oxygen depletion (every late summer and autumn), the number of benthic species is reduced significantly and mature communities cannot develop. This phenomenon can be compared with forest fires on land. If a particular area is struck with fire too often, new trees will not have a chance to reach maturity and it may be impossible to establish stable and widespread forests again. In marine areas with temporary oxygen depletion, intermittent recovery will occur whenever conditions improve. In this process, it is usually small and rapidly reproducing species with good dispersal ability that dominate.

Oxygen depletion may be viewed as a temporal and spatial mosaic of disturbance that results in the loss of habitats, reductions in biodiversity, and a loss of functionally important species. In a Baltic-wide perspective, these disturbances have also resulted in a reduction in the connectivity of populations and communities, which impairs recovery potential and threatens ecosystem resilience. Recovery of benthic communities is scale-dependent and an increase in the extent or intensity of hypoxic disturbance may dramatically reduce rates of recovery.

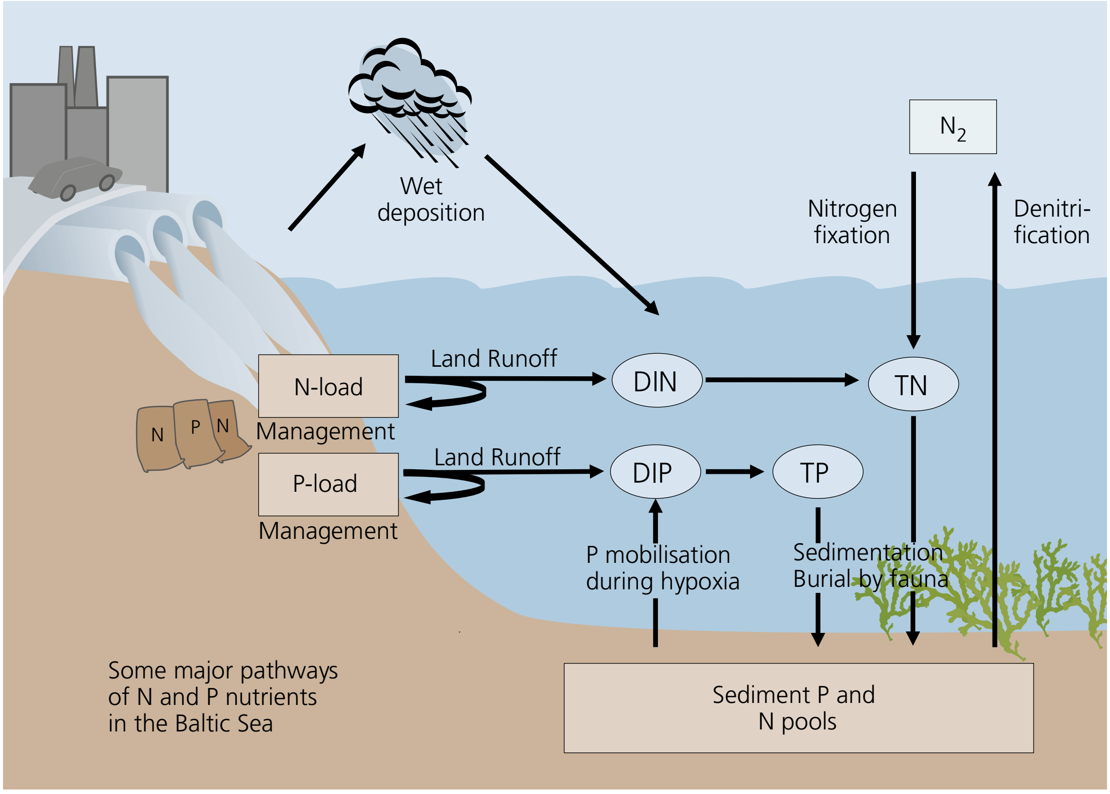


Figure 2.3 Simplified conceptual model for N and P nutrients in the Baltic Sea, where DIN = Dissolved inorganic nitrogen, TN = Total nitrogen, DIP = Dissolved inorganic phosphorus and TP = Total phosphorus. Flows along arrows into the blue sea area tend to increase concentrations, and flows along arrows out from the sea act in the opposite direction. Management refers to nutrient load reductions.

Benthic communities, both in soft sediments and on hard substrates, are often composed of important habitat-forming species. Such healthy, diverse benthic communities provide important ecosystem services, including the provision of food for higher trophic level organisms and the mineralization of settling organic matter. It is evident that reductions in the distribution and diversity of benthic macrofauna, owing to hypoxic events, have severely altered the way benthic ecosystems contribute to ecosystem processes in the Baltic Sea.

A comprehensive assessment of the status and trends of benthic invertebrate communities in the Baltic Sea is contained in **Chapter 2.7**.

2.2 Nutrients

The external loads of nutrients to the Baltic Sea come principally from land and from atmospheric deposition, but internal loads from sediments and the fixation of atmospheric nitrogen by cyanobacteria can also be substantial (**Fig. 2.3**; Savchuk & Wulff 2007). The major removal pathways of nutrients are through permanent sediment burial, denitrification (only N), and export to the Skagerrak.

Marine eutrophication is mainly caused by nutrient enrichment leading to increased production of organic matter (Nixon 1995) supplied to the Baltic Sea with subsequent effects on water transparency, phytoplankton communities, benthic fauna and vegetation as well as oxygen conditions (see **Chapter 2.6**). Phytoplankton need nutrients, mainly nitrogen and phosphorus, for growth, ideally according to the Redfield ratio (N:P=16:1 on a molar basis). Silica (Si) is also needed for the most common phytoplankton group, diatoms, but it is naturally available in large amounts in the Baltic Sea. Low concentrations of bioavailable N and P will limit primary production, and both N and P can be limiting exclusively or in combination (co-limitation). Ambient concentrations of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) are often used to assess potential nutrient limitation, with the suggestion that primary production is mainly N-limited for DIN:DIP below 10 and mainly P-limited for DIN:DIP above 20. Bioassays have shown that primary production is mostly P-limited in the Bothnian Bay (Andersson et al. 1996; Tamminen & Andersen 2007) and mostly N-limited in the Kattegat (Granéli et al. 1990), but nutrient limitation patterns switch during seasons (Tamminen & Andersen 2007), in relation to proximity to freshwater sources (Pitkänen & Tamminen 1995), and during blooms of cyanobacteria (Lignell et al. 2003; Nausch et al. 2004). Mitigation of eutrophication effects therefore needs to address both N and P.

Seasonal variations in supply, removal, and transformation processes give rise to distinct seasonal patterns for nutrient concentrations, which are most pronounced for DIN and DIP that often become depleted in surface waters during summer. Distinct spatial gradients are also observed, with elevated nutrient concentrations in estuaries and coastal waters compared to open waters. These seasonal and spatial variations must be taken into account when assessing the trend from heterogeneously sampled monitoring data. This has been achieved for the status and trends reported here, using the statistical approach from Carstensen et al. (2007), where each observation is weighted according to location and month of sampling before computing annual means, with the restriction however that locations with fewer than 20 observations were not used. Status and trends are presented as geometric means.

2.2.1 Status 2001–2006

Nutrient status for the period 2001–2006 was calculated for inorganic nutrients as winter means (December–March) and for total nutrients as annual means (January–December) using the same generic approach (Carstensen et al. 2006) as for the trend assessment with the modification that a single mean is calculated for all years 2001–2006 as opposed to six means. In this assessment, values have been compared for nutrient status in surface water (0–10 m) between the different basins (**Fig. 2.4**).

The highest DIN concentrations were in the Bothnian Bay, which is predominantly P-limited and therefore DIN may accumulate to reach levels above those in other basins. DIN concentrations in the Gulf of Finland were also high owing to large loads of nutrients mainly from the Neva River. For the other basins, DIN winter concentrations varied between 3 and 4 µmol l^{-l}. The Gulf of Riga and Gulf of Finland had the highest TN annual concentrations (26 and 24 µmol l^{-l}, respectively), which are due to large riverine discharges to both basins. The other basins had TN concentrations between 18 and 21 µmol l^{-l}, with the lowest concentrations in the Danish Straits. From the Baltic Proper to the Danish Straits, there is a natural decreasing spatial gradient owing to the mixing with Skagerrak surface water that generally has lower TN levels.

High DIP winter concentrations were found in the Gulf of Riga and the Gulf of Finland (0.78 and 0.84 µmol l^{-l}, respectively) owing to the large influence from riverine discharges and the mixing of bottom waters rich in phosphorus deriving from the Baltic Proper (Pitkänen et al. 2001). DIP concentrations in the Bothnian Sea, Baltic Proper and Danish Straits were similar (0.35 to 0.47 µmol l^{-l}), whereas DIP concentrations in the Bothnian Bay were very low (0.06 µmol l^{-l}). These spatial differences were unaltered for TP, with high levels in the Gulf of Riga and Gulf of Finland (0.70 and 0.85 µmol l^{-l}, respectively), moderate TP levels in the Baltic Proper and Danish Straits (~0.58 µmol l^{-l}) with slightly lower levels in the Bothnian Sea (0.42 µmol l^{-l}) and substantially lower in the Bothnian Bay (0.16 µmol l^{-l}).

2.2.2 Temporal trends

Time series of nutrient concentrations in surface (0–10 m) and bottom waters (> 100 m) starting from the 1970s until 2006 have been analysed to derive both winter (December–March) and annual (January–December) means using the statistical approach from Carstensen et al. (2006). The advantage of using annual means relative to winter means is that more precise indicators are obtained, provided that the seasonal variation is accounted for (Carstensen 2007). For the presentation of the results, the Kattegat and Belt Sea as well as the Bothnian Sea and Archipelago Sea have been merged in order to limit the number of plots.

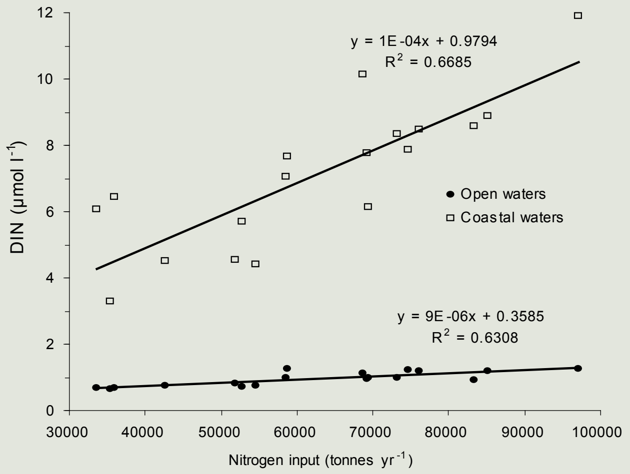
Surface water

In this report, only winter means of inorganic nutrients in the surface layer (**Fig. 2.5**) and annual means of total nutrients and their ratio (**Fig. 2.6**) have been shown. Additional time series plots are presented in HELCOM (2009). The nutrient

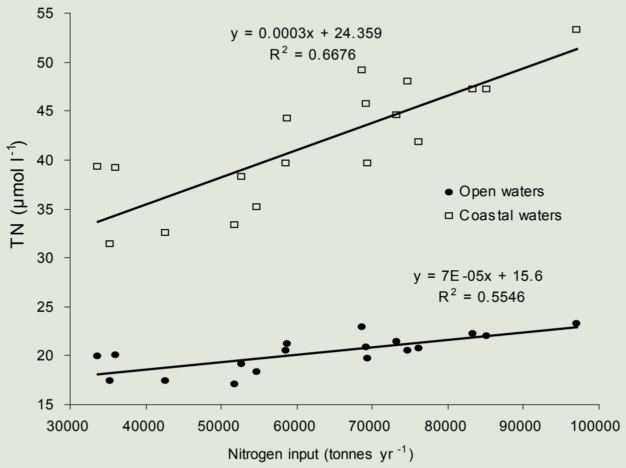
BOX 1: Correlations between nutrient concentrations and loads

Increasing discharges of nutrients to a water body will, among others, result in an increase in nutrient concentrations. This is very well-documented for lakes and many enclosed marine systems such as estuaries and fjords. In these systems, loss mechanisms are of minor importance compared to the magnitude of the inputs.

In coastal waters and in large and complex estuarine systems such as the Baltic Sea, the relationship between loads and nutrient concentrations is not as simple as for the enclosed systems. Loss mechanisms (sedimentation, denitrification) and retention time play key roles but obscure the cause-effect relationships. However, in the Kattegat and Danish Straits, the retention is rather low compared to the retention times of other parts of the Baltic Sea, especially the Baltic Proper. As a consequence, the relationship between loads and nitrogen concentrations is significant, especially for the coastal waters. Panels A and B illustrate the functional relationship between nitrogen loads and DIN as well as TN concentrations. The key messages are: (1) increased loading leads to nutrient enrichment in both coastal and open waters, and (2) reduced loadings will result in a decline of nutrient concentrations as well an alleviation of eutrophication signals.



Panel A: Annual mean DIN concentrations estimated from the Danish monitoring data versus nitrogen input from land (1989–2006).



Panel B: Annual mean TN concentrations estimated from the Danish monitoring data versus nitrogen input from land (1989–2006).

trends have also been analysed together with a preliminary loading compilation (1994–2006) to indicatively assess the extent to which local sources affect nutrient concentrations, but the plots are not shown and detailed statistics not presented because the loading data still need to be validated.

Bothnian Bay

TN concentrations in the open waters increased from 15 µmol l^{-l} in the beginning of the 1970s to about 20 µmol l^{-l} in the mid-1980s and have stabilized at this level since then. A similar trend was observed for DIN, with increases from 6 µmol l^{-l} to 7–8 µmol l^{-l}, although this tendency was not as clear for winter means owing to limited data. Nitrogen levels in the open sea showed no response to nitrogen loading from land, despite considerable year-to-year variations (factor of 2). This suggests that nitrogen levels are governed more by exchanges with the Bothnian Sea, which shows similar nitrogen trends. Nitrogen concentrations in the coastal zone were elevated compared to the open sea and more variable, but no long-term trend could be discerned owing to a lack of data. In the coastal zone, nitrogen levels showed a weak positive correlation with local loads, suggesting that land-based loads can have a direct impact on nitrogen levels but exchanges with the open sea are also important.

TP concentrations in the open waters increased slightly up to 1980 but then decreased, and a similar pattern was observed for DIP, declining from ca. 0.1 to 0.06–0.07 µmol l^{-l}. Similar to nitrogen, phosphorus levels in the open waters were not related to land-based loads, suggesting that exchanges with the Bothnian Sea and internal processes are more important. In the coastal zone, TP levels have decreased from 0.7 µmol l^{-l} in the late 1970s to less than 0.4 µmol l^{-l} in recent years. Phosphorus levels in the coastal zone were correlated with loads from land, indicating that local sources directly affect P concentrations in the coastal zone. The decreasing P levels have increased the N:P ratio, but there is an overall excess of N relative to P, supporting the current findings that the Bothnian Bay is generally P-limited.

Silica concentrations in the open sea and coastal zone were similar, with a tendency to increasing levels since 1980. This trend could be explained by the decreasing DIP concentrations binding less dissolved inorganic silica (DSi) in diatom biomass. DSi concentrations in the Bothnian Bay are generally high relative to DIN and DIP and therefore silica limitation of diatoms is unlikely.

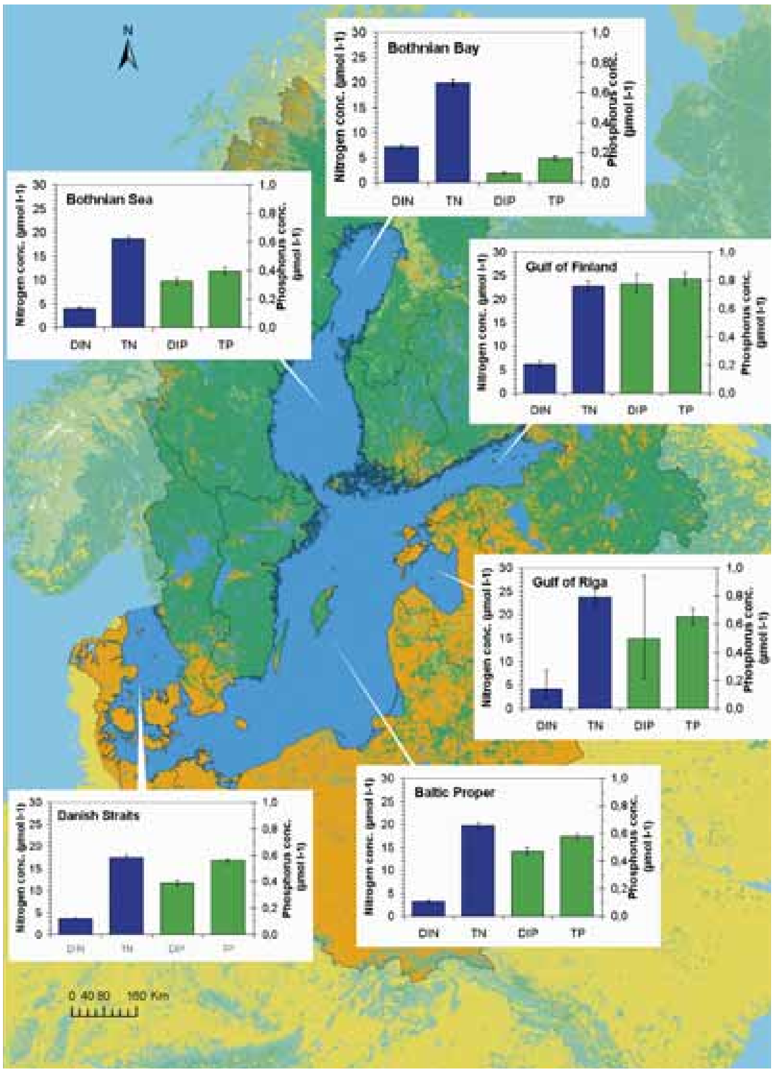


Figure 2.4 **Map of the Baltic Sea area with assessments of four nutrient components in open parts of the six sub-areas: Bothnian Bay, Bothnian Sea, Gulf of Finland, Gulf of Riga, Baltic Proper and the Danish Straits. DIN and DIP represent winter means, and TN and TP mainly represent annual means.**

Bothnian Sea

TN in the open sea showed a trend similar to that the Bothnian Bay, with increases from 15 µmol l^{-1} in the early 1970s to 20 µmol l^{-1} in the mid-1980s. DIN also increased slightly during the same period from 4 to 5 µmol l l^{-1}. DIN levels in the Bothnian Sea were consistently lower than those in the Bothnian Bay, supporting the general change from P-limitation to N-limitation, which has been documented by HELCOM (2006b) and Tamminen & Andersen (2007). Nitrogen levels in the open Bothnian Sea were slightly more correlated to local loads than in the Bothnian Bay, but the relationships were not strong. DIN and TN in the coastal zone followed the same trend as for the open sea with concentrations ca. 3 and 5 µmol l^{-1} higher, respectively. N levels in the coastal zone were positively, but not significantly, correlated with loads from land. The weak relationship and slightly elevated N concentrations in the coastal zone suggest that exchanges with the open sea are the most important mechanism governing N levels.

TP concentrations increased from 0.3 µmol l^{-1} in the early 1970s to about 0.5 µmol l^{-1} in the mid 1980s, with a weak tendency to decrease since then. DIP also showed increases during the same period from less than 0.3 µmol l^{-1} to 0.4 µmol l^{-1}, with a stabilization around 0.35 µmol l^{-1} in the most recent years. Similar to N, there were positive correlations between P loads and concentrations, but they were not significant. In the coastal zone, both the increase and decline of TP levels were stronger than for the open sea, magnify-

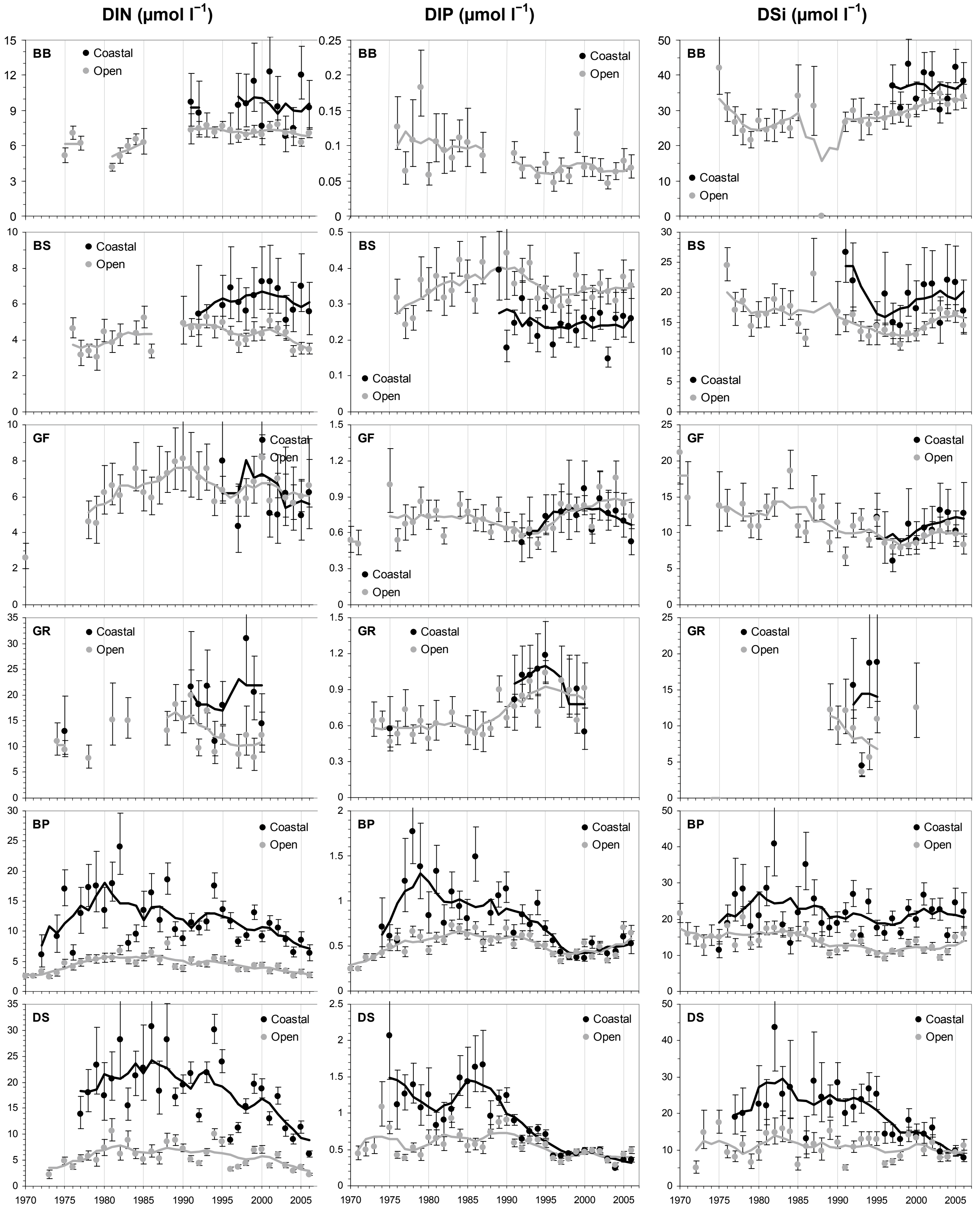


Figure 2.5 Winter means (December–March) of dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), and dissolved inorganic silica (DSi) in surface water (0–10 m). Error bars show the 95% confidence limits of the means. Solid curves are 5-year moving averages. Means with a relative uncertainty larger than 25% are not shown. BB = Bothnian Bay; BS = Bothnian Sea; GF = Gulf of Finland; GR = Gulf of Riga; BP = Baltic Proper; DS = Danish Straits.

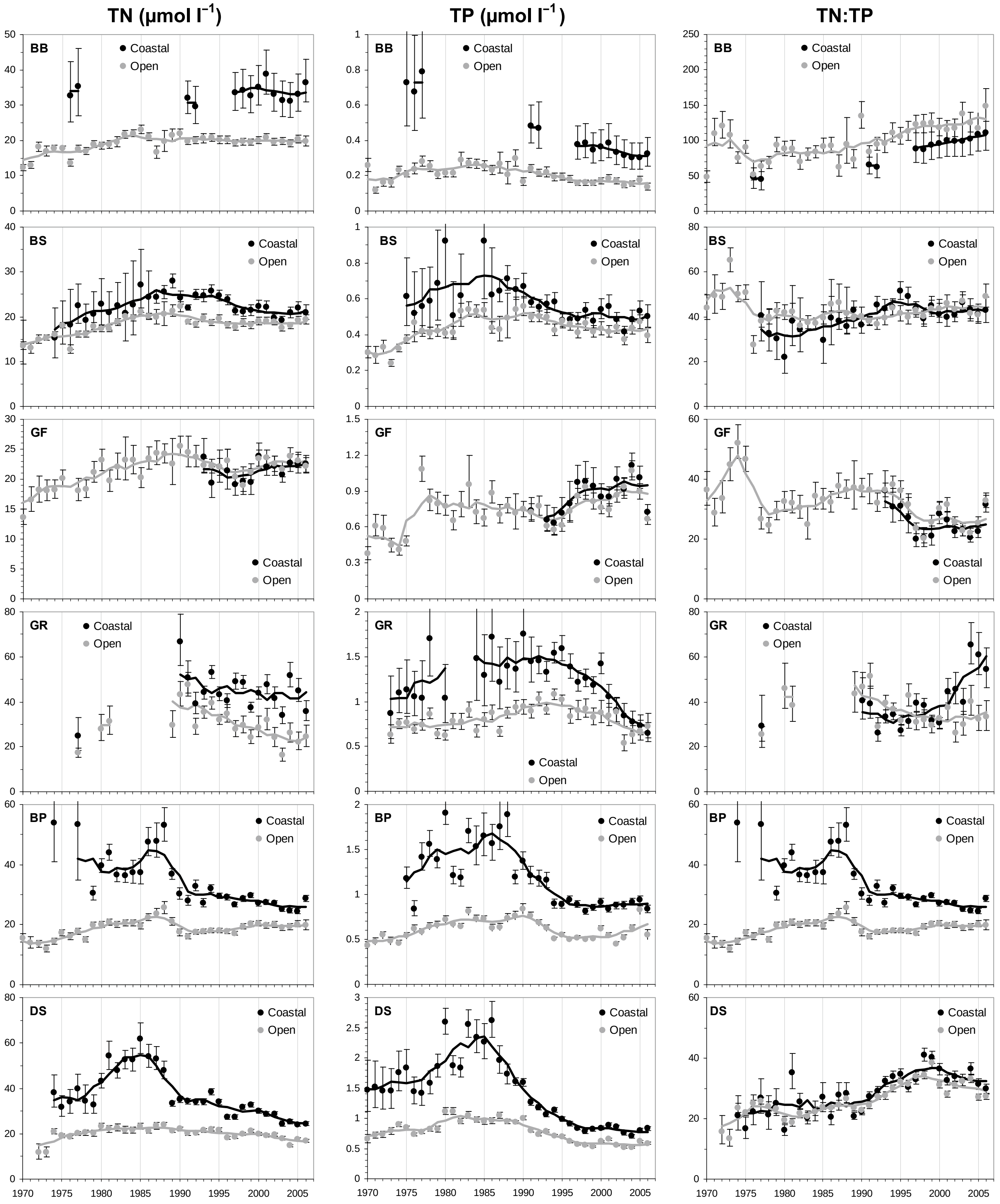


Figure 2.6 Annual means (January–December) of total nitrogen (TN), total phosphorus and their ratio (TP) in surface water (0–10 m). Error bars show the 95% confidence limits of the means. Solid curves are 5-year moving averages. Means with a relative uncertainty larger than 25% are not shown. BB = Bothnian Bay; BS = Bothnian Sea; GF = Gulf of Finland; GR = Gulf of Riga; BP = Baltic Proper; DS = Danish Straits.

ing the overall tendencies. DIP levels were lower in the coastal zone than in the open sea, which is probably due to switching from P-limitation in the coastal zone to N-limitation in the open sea. The winter DIN:DIP ratio has decreased over the past ten years suggesting a switch towards more co-limitation in the coastal zone and stronger N-limitation in the open sea.

Silica concentrations declined from the 1970s to the mid-1990s, with an increase since then. These trends are probably due to changes in DIN and DIP levels and discharges from land. DSi levels are still relatively high compared to DIN and DIP, indicating that silica limitation of diatom growth is unlikely.

Gulf of Finland

TN concentrations increased by almost 10 µmol l^{-1} from 1970 to the mid-1980s in the open sea, followed by a down-up trend. DIN concentrations displayed more or less the same tendency, although they were more variable. N levels in the coastal waters were almost identical to those in open waters, suggesting that exchanges with the open sea dominate the coastal waters. The relationships between N levels and loads from land could not be evaluated owing to inconsistent load data.

TP concentrations were low in the 1970s (~0.5 µmol l^{-1}) jumping to higher levels in the 1980s (~0.8 µmol l^{-1}), followed by a decreasing trend and then increasing since 1990. DIP concentrations similarly reached a low around 1990, rising to higher values most recently. The trends in P appear to be linked with the trends in the Baltic Proper, whereas the effect of changing P loads from land, particularly those in the Neva River, cannot be assessed. The different trends for DIN and DIP have had ramifications for the DIN:DIP ratio (winter) that increased from about 7 in the 1970s to almost 15 in the 1980s and then returned to about 7 again recently. With these temporal changes in the N:P ratio, the boundary between the N- and P-limited areas has also moved west and then east again.

Silica concentrations declined steadily from 1970 to 1995, almost halving the levels in both the open sea and the coastal zone. In recent years, DSi has increased slightly again to reach levels around 10 µmol l^{-1}. These trends are believed to be related to nutrient enrichment from land in the first period binding more silica in biomass, and to increasing loads from the Baltic Proper in the second period. DSi winter levels are only slightly higher than DIN and DIP according to Redfield, suggesting that silica limitation could be present.

Gulf of Riga

Data are scarcer from this region compared to the other regions assessed. In particular, winter nutrient levels could only be determined with sufficient precision in a limited number of years.

TN concentrations in both open and coastal waters have declined by 20–40% since the mid-1980s, attaining levels presumably present in the early 1970s. DIN concentrations have similarly declined since the mid-1980s in the open sea, but the tendency is not clear for coastal waters owing to a lack of data. Recent N levels in coastal waters are almost twice as high as levels in the open sea, indicating strong N gradients towards land. N levels in both the coastal zone and open sea were strongly related to N loads from land, stressing the importance of land-based loads to this region.

TP concentrations gradually increased from the 1970s up to 1990 and then rapidly declined to almost half the maximum levels for both coastal and open waters. Maximum levels around 1990 were 1.5 and 1.0 µmol l^{-1} for coastal and open waters, respectively. DIP levels also increased until around 1990 and then declined by almost 50%, most pronounced in the coastal zone. These conspicuous declines were apparently not related to changes in the P load from land because TP loads were quite stable from 1994–2006 at around 1,500–3,000 tonnes yr^{-1}. The observed trends can only partly be explained by changes in P levels in the Baltic Proper; therefore, internal loads could also be significant.

Silica concentrations (annual means) have gradually increased since the 1990s, most likely in response to decreasing N and P levels and therefore excess silica. Winter DSi levels around 1990 were, however, comparable to DIN and DIP levels assessed by the Redfield ratio, suggesting that silicate limitation might have been present in those years. Potential silicate limitation has probably been relieved in recent years owing to decreasing DIN and DIP concentrations.

Baltic Proper

TN concentrations in the open waters increased from about 15 µmol l^{-1} in the early 1970s to above 20 µmol l^{-1} in the mid-1980s, then quickly dropped down to ca. 17 µmol l^{-1} around 1990; since then TN has steadily increased to the present levels of just below 20 µmol l^{-1}. Winter DIN, on the other hand, almost doubled from 1970 to the mid 1980s (~6 µmol l^{-1}) and then steadily declined back to around 3 µmol l^{-1}. This discrepancy between winter DIN and annual TN is probably due to increased nitrogen fixation during the summer period, which is also reflected in that winter DIN levels are related to loads from land whereas annual TN shows no relationship at all. The uplift of the halocline after the major saltwater inflows in 1993 and 2003 resulted in a mixing of waters containing low DIN concentrations through the halocline (Nausch et al. 2003). In the coastal zone, the trends of DIN and TN are more comparable with increasing trends up to the mid-1980s, followed by declines. Thus, nitrogen fixation appears only to affect annual TN trends in the open sea. For the same reason, significant correlations were obtained between N loads from land and N levels in the coastal zone.

TP concentrations in the open sea steadily increased from 1970 up to 1990, followed by a 0.3 µmol l^{-1} drop in the early 1990s, and then increased again recently. Trends for DIP were quite similar. These trends correspond to the dynamics of the deep water renewal in the Baltic Proper, where large pools of DIP were entrained into the surface layer during the long stagnation period through a gradual deepening of the halocline (Conley et al. 2002a). The major inflows to the Baltic Proper in 1993 and 2003 caused a halocline uplift, increasing the potential for entraining large pools of DIP into the surface layer. Thus, recent P levels in the open sea are governed by P mixing across the halocline. DIP and TP concentrations in the coastal zone were considerably higher than the levels in the open sea in the 1970s and 1980s, but they have been more comparable with levels open waters in the past two decades with regard to both magnitude and trends. This suggests that P levels before 1990 may have been affected by land-based loads, whereas more recent P levels appear linked to the dynamics of the open sea. The further decreasing winter DIN:DIP ratio after 1993 indicates an N-limitation of the spring phytoplankton bloom.

Silica concentrations in the open sea show small decreases from 1970 to around 1995, followed by small increases; coastal waters exhibit comparable trends. It is most likely that, as for P, these trends are related to the entrainment of bottom waters into the surface layer. Comparing inorganic nutrient winter levels shows excess DSi and therefore silica limitation is unlikely.

Danish Straits

TN concentrations in the open sea increased from around 15 µmol l^{-1} in the early 1970s to about 22–23 µmol l^{-1} in the mid-1980s. Action plans in the region to reduce nutrient loads (Carstensen et al. 2006) have contributed to reduce TN levels down to 16–17 µmol l^{-1} in the most recent years. Similar trends are observed for DIN, where present levels are comparable to those in the 1970s. N levels in coastal waters in the 1980s were the highest of all the regions assessed in this report, and 3–4 times higher than the levels in the open sea, indicating the large influence of land-based sources. Since the mid-1980s, N levels in coastal waters have decreased to approximately half, with differences between coastal and open sea levels dropping to a factor of 2. Significant relationships between loading and concentrations (Carstensen et al. 2006, this assessment) document a strong link to land-based loads on a year-to-year basis due to the low retention times.

DIP and TP concentrations in the open sea also increased from 1970 to the mid-1980s, with subsequent declines reaching levels that are now below the levels in 1970. P levels in coastal waters were also the highest of all regions in the mid-1980s, but DIP levels are now comparable to those in open waters and TP levels are only slightly above. The significant declines in P levels primarily result from deliberate management actions to reduce loads from point sources (Carstensen et al. 2006), and P levels are significantly related to the loads for both coastal and open waters.

Silica concentrations in the open waters have remained more or less constant at around 10 µmol l^{-1} for the past four decades. Winter DSi levels have decreased since the 1980s, and one explanation for this could be that measures to reduce nutrient loads from diffuse sources have also had an impact on soil erosion. DSi levels are relatively higher than DIN and DIP according to the Redfield ratio, and silica limitation is not likely to be pronounced.

Bottom water

Inorganic nutrient trends (annual means) for bottom water were assessed only for the Bothnian Bay, Bothnian Sea, and Baltic Proper (**Fig. 2.7**). A typical sampler used to collect water for nutrient measurements is shown in **Fig. 2.8.** Trend plots of total nutrients (annual means) can be found in HELCOM (2009). Only the western part of the Gulf of Finland is deep enough to have bottom waters below the permanent halocline, but the properties of these waters are similar to those in the northern Baltic Proper. The Gulf of Riga does not have permanently stratified waters throughout the entire basin, so properties in the deeper waters are similar to those in the surface layer. In the Bothnian Sea and especially the Bothnian Bay, there is no permanent halocline and the water column mixes all the way to the bottom. The Danish Straits have a permanent halocline located around 15 m depth, but nutrient trends in the surface and bottom layer are similar for this region owing to intense up- and downward mixing (Carstensen et al. 2006).

Bothnian Bay

DIN levels increased from 6–7 µmol l^{-1} in the 1970s to above 8 µmol l^{-1} in the 1990s, with a tendency to lower levels in recent years. Nitrogen trends in bottom water were similar to those in the surface water. Annual DIP levels were about twice the levels in the surface layer, but the overall trends were similar, as they were also for TP. DSi levels mostly remained close to around 30 µmol l^{-1}, with some lower levels around 1980 and 1990, which were only partly reflected in the surface levels. Overall, bottom concentrations appear to be linked to processes in the surface layer.

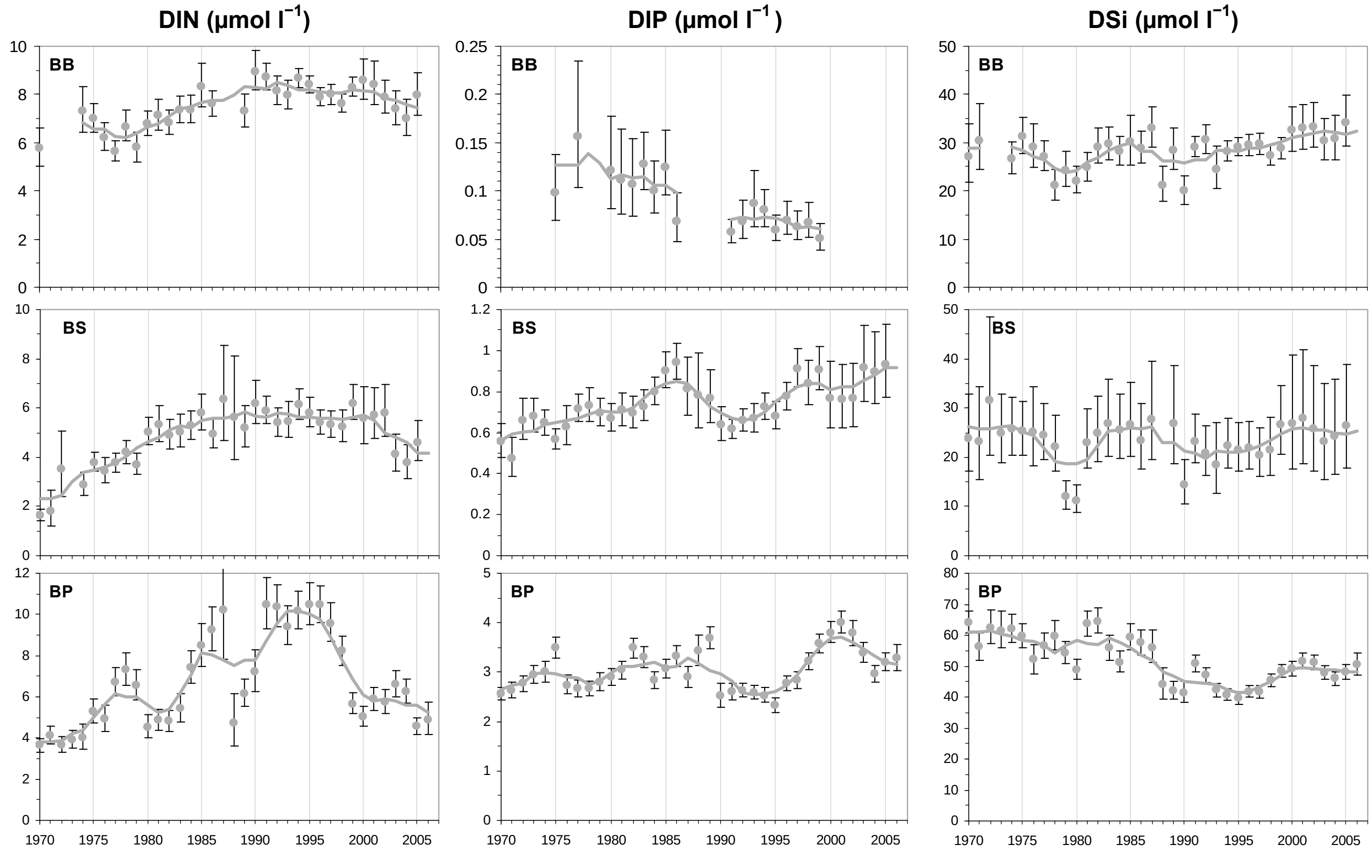


Figure 2.7 Annual means (January–December) of dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), and dissolved inorganic silica (DSi) in bottom water (> 100 m). Error bars show the 95% confidence limits of the means. Solid curves are 5-year moving averages. Means with a relative uncertainty larger than 25% are not shown. BB = Bothnian Bay; BS = Bothnian Sea; BP = Baltic Proper.

Bothnian Sea

DIN increased drastically from about 2 µmol l^{-1} in 1970 to almost 6 µmol l^{-1} around 1990, where the level stabilized. For the past three years, lower DIN values have been observed. DIP values have increased over time from 0.6 µmol l^{-1} in 1970 to 0.9 µmol l^{-1} in recent years. The trend showed a cyclic behaviour during the 1980s and 1990s, which may be related partly to trends in surface water and partly to exchanges with the Baltic Proper. DSi levels were rather stable at around 25 µmol l^{-1}. Overall, mixing with the surface layer and exchanges with the Baltic Proper appear to be the most important processes for bottom nutrient concentrations.

Baltic Proper

DIN concentrations more than doubled from 1970 to 1995 and then decreased by 50% within a few years. In addition, oscillations in DIN levels of almost decadal scale were observed. The trends in DIN were governed by bottom water renewal processes through major inflows and enhanced denitrification associated with an increasing volume of hypoxia (Conley et al. 2002a, Vahtera et al. 2007). Before the major inflow in 1993, the hypoxic water volume was at its lowest yielding an overall lower DIN removal rate. DIP was also affected by changing volumes of hypoxia, because DIP is released from the sediments when exposed to hypoxia. DSi concentrations gradually declined from 60 µmol l^{-1} in 1970 to 40 µmol l^{-1} around 1995 and then increased to about 50 µmol l^{-1} in recent years. Although silica processes are not directly redox dependent, remineralization processes in the sediments may occur at different rates during oxic, anoxic and anaerobic conditions. Overall, nutrient concentrations in the bottom waters are governed by changes in the large volume of hypoxic water modulating the biogeochemical processes in both the sediments and water column.

Figure 2.8 **Sampling of water for analysis of nutrients.**

2.3 Phytoplankton

Phytoplankton respond rapidly to changes in nutrient levels and, therefore, the biomass and species composition of phytoplankton can be used as indicators of eutrophication (HELCOM 2006). This section reports the state and trends using the phytoplankton indicator that has been chosen to define reference conditions, namely, the chlorophyll-*a* concentration. In addition, the state and trends for a set of other phytoplankton indicators are also presented.

Phytoplankton are important primary producers in the coastal and open Baltic Sea and they provide energy for the higher components of the food web. The socio-economic importance of phytoplankton is associated with algal blooms and their potential toxicity, which reduce the recreational use of the water and pose a health risk to humans and aquatic organisms (Edler et al. 1996; GEOHAB 2001; Sivonen et al. 2007; Karjalainen et al. 2007; Uronen 2007). Phytoplankton blooms decrease water transparency (**Chapter 2.4**) and light availability, which reduces the living conditions for submerged vegetation in the coastal areas (**Chapter 2.5**); blooms also increase the sedimentation of organic material, which enhances oxygen con-

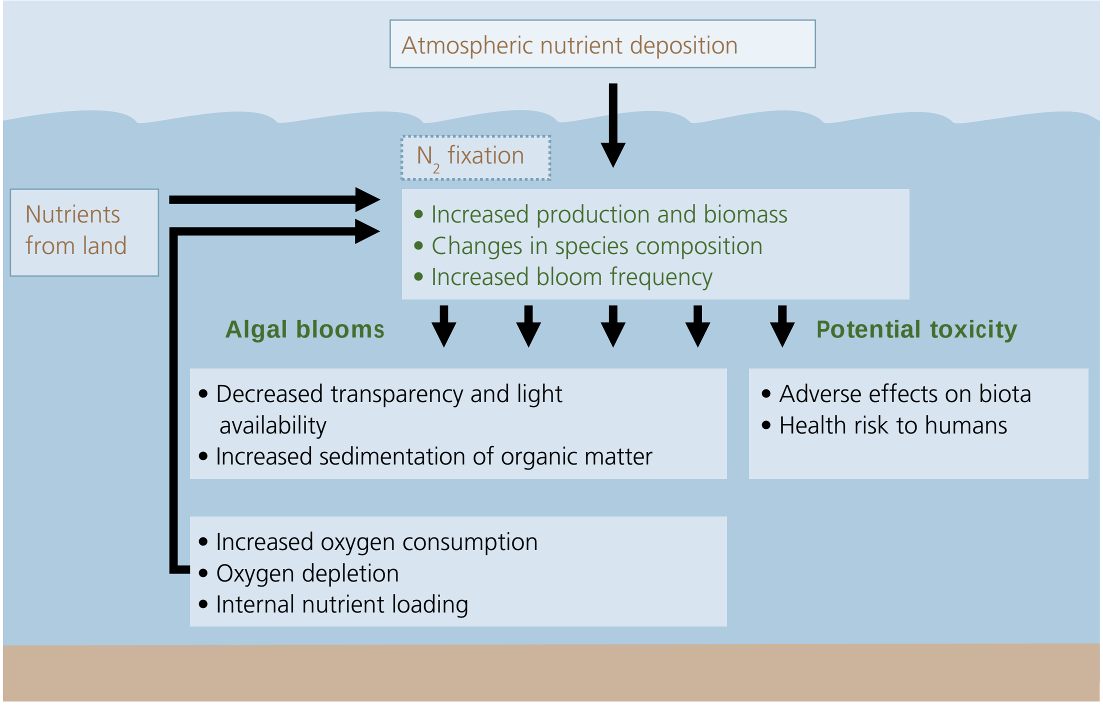


Figure 2.9 Conceptual model of the relationship of phytoplankton to eutrophication in the Baltic Sea.

sumption in the near-bottom waters and internal nutrient loading (**Chapters 2.2 & 2.6**) (**Fig. 2.9**).

Phytoplankton biomass and communities in the Baltic Sea reflect both hydrological conditions and the eutrophication process (HELCOM 2002, Wasmund & Uhlig 2003). Phytoplankton cells derive nutrients directly from the water column, and under the favourable conditions they can grow and reproduce rapidly. The close connection between phytoplankton and nutrient levels makes phytoplankton a useful tool in the assessment of the state of aquatic ecosystems.

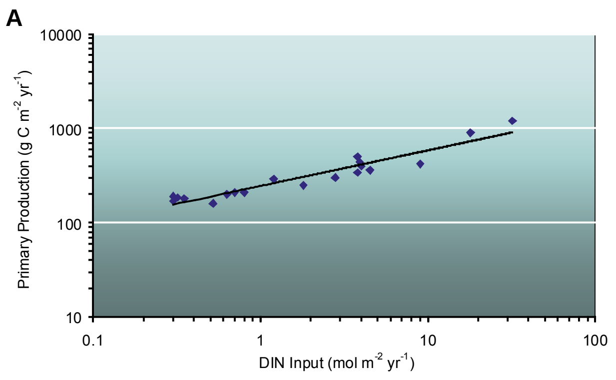
Altogether eleven different phytoplankton indicators were developed and proposed for use in the eutrophication assessment of the Baltic Sea (HELCOM 2006b; Kuuppo 2007). The indicators proposed are intended to be practical and measurable; however, especially many of the phytoplankton species indicators do not perfectly meet these criteria (HELCOM 2006b; Kuuppo 2007). An uneven geographical distribution also complicates the use of certain phytoplankton groups in the assessment of the entire Baltic Sea region (Carstensen et al. 2004a; Gasiūnaitė et al. 2005).

At present, reference conditions have been determined only for chlorophyll-*a* concentrations in summer and tentatively for the abundance of the cyanobacterium *Aphanizomenon flos-aquae* in summer; reference conditions for the other indicators are currently under development (HELCOM 2006b; Kuuppo 2007; Fleming-Lehtinen et al. 2007a). Hence, this report focuses mainly on chlorophyll-*a* in summer (June–September) in different sub-basins of the Baltic Sea.

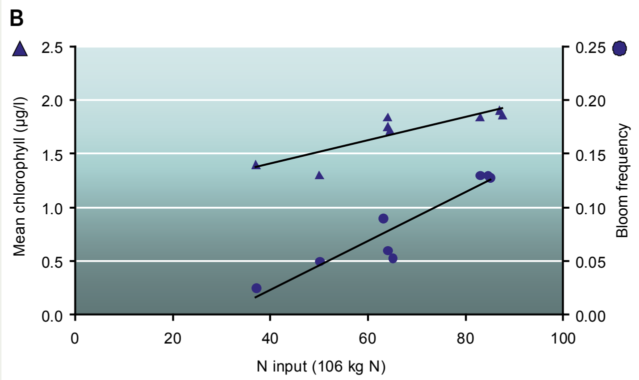
The chlorophyll-*a* concentration is a commonly used proxy of phytoplankton biomass, because all phytoplankton cells contain chlorophyll-*a* as their main photosynthetic pigment. Chlorophyll-*a* is measured routinely in monitoring programmes (see **Fig. 2.10**, for common sampling methods), and it reflects total phytoplankton biomass relatively well. The analysis is widely and commonly used as a standardized method, and there are several long-term time series data of chlorophyll-*a* available. In the Baltic Sea, chlorophyll-*a* is also routinely monitored with automated flow-through sampling on merchant ships (Fleming-Lehtinen & Kaitala 2006a). In addition, surface blooms of phytoplankton (as chlorophyll-*a*) are monitored in open waters using satellite remote sensing data (Schrimpf & Djavidnia 2006; Hansson 2007), which provides information with large spatial coverage.

BOX 2: Correlations between chlorophyll-*a* concentrations and nutrient loading

Phytoplankton primary production and biomass are closely related to nutrient inputs in the Baltic Sea and other marine environments. It is often generalized that nitrogen limits phytoplankton in the open and coastal waters of the Baltic Sea. However, depending on the area and season also phosphorus, and in the case of diatoms, silica can be a limiting nutrient for phytoplankton growth. An increase in a nutrient that is limiting for primary production will promote primary production more than increases in any other nutrients. Clear dose-response relationships between nutrient load and phytoplankton biomass are modified, e.g., by top-down control of zooplankton.



Panel A: Annual phytoplankton primary production in relation to the estimated rate of input of dissolved inorganic nitrogen (DIN) per unit area in different marine ecosystems. Redrawn from Nixon et al. (1996).



Panel B: The relationship between annual nitrogen inputs (106 kg N) and mean summer chlorophyll-*a* concentration and bloom frequency in the Kattegat area. Redrawn from Carstensen et al. (2004b).

2.3.1 Status 2001–2006

Similar to water transparency (**Chapter 2.4**), the assessment of coastal and transitional waters using summer (June–September) chlorophyll-*a* concentrations is mostly based on the work carried out by the national experts from HELCOM Contracting Parties. Chlorophyll-*a* data from several stations are integrated to obtain the assessment for each region. Harmonization of the chlorophyll-*a* methods, including the reference conditions, has been conducted in connection with the implementation of the EU Water Framework Directive and the HELCOM EUTRO project. The chlorophyll assessment focuses on the time period 2001–2006, but for the Latvian waters the period is 2001–2004/2005.

The assessment of the open Baltic Sea is based on the HELCOM COMBINE chlorophyll-*a* data (HELCOM 2006). Reference conditions for the Bothnian Bay and the Gulf of Finland are defined in Fleming-Lehtinen (2007); for the other open sea areas, the reference values suggested by EUTRO-PRO and DHI are used (HELCOM 2006). The reference values used are largely in line with those presented in Schernewski & Neumann (2005). For Polish waters, reference conditions for the annual chlorophyll-*a* means were used along with the summer reference values.

In the open Baltic Sea, between 2001 and 2006 the chlorophyll-*a* based EQR (Ecological Quality Ratio) values varied from 0.22–0.67 in summer (June–September) (**Fig. 2.11**). The lowest status expressed as EQR values – indicating substantial deviations from reference condition values – were found in the Gulf of Finland, the Northern Baltic Proper and the Gulf of Riga. The highest EQR values – indicating no or slight deviations from reference condition values – were found in the Bothnian Bay and the Kattegat. In the Arkona Basin, Bornholm Basin and different Gotland Basins, the EQR values ranged from 0.40 to 0.55. Depending on the sub-basin, the EQR values were equivalent to increases from 33% to 78% in the average summer chlorophyll-*a* levels in relation to reference conditions.

Figure 2.10 Sampling of water for analysis of phytoplankton (panel A) and chlorophyll-*a* (panel B) of an algal bloom.

In the coastal and transitional waters, the status expressed as EQR values also showed geographical variation (**Fig. 2.12**). The EQR values in many coastal areas were higher in the southern parts of the Baltic Sea than in the respective northern areas. In the Kattegat, the EQR values were generally high and varied from 0.37 to 0.81, i.e. from substantial deviations from reference conditions to slight deviations. The EQR values were higher in the outer than in the inner coastal waters of the Kattegat. In the northern and central Sound, the EQR values were at the same level (0.59 and 0.54, respectively), whereas the status was higher in the southern Sound (EQR 0.93).

In the Belt Sea coastal waters, the EQR values varied from 0.30 to 0.73, with the highest status in the Kiel Bight (0.73). The status of the Arkona inner and outer coastal waters was 0.80 and 0.70, respectively. The Fehmarn Belt, Lübeck Bight and Darss-Zingst outer coastal waters showed very similar EQR values (0.78–0.85), whereas the status was lower in the Wismar Bight (0.37). In the Mecklenburg Bight, the status expressed as an EQR value was 0.57.

In the Bornholm Basin, the chlorophyll-based EQR values in the outer and inner Pommeranian Bay were 0.47 and 0.24, respectively, and very high (1.0) in the Hanö Bight. In the Gulf of Gdańsk, the EQR values were rather similar in the Vistula mouth (0.43), the Outer Puck Bay (0.35), and the transitional waters (0.43). In the Lithuanian north-

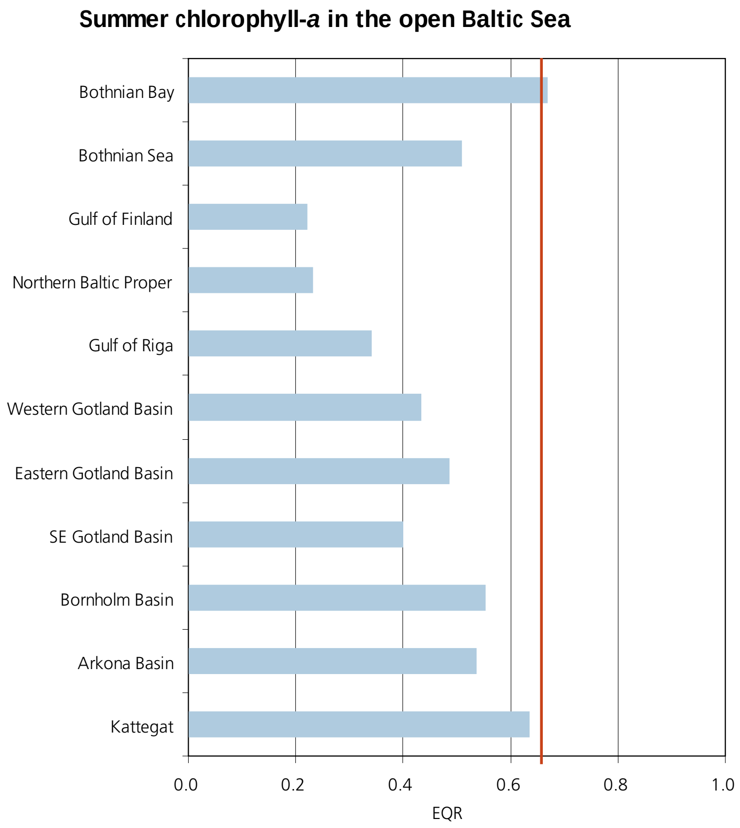


Figure 2.11 Chlorophyll status in Baltic Sea open areas expressed as Ecological Quality Ratio (EQR) values. The EQR values are based on the average summer (June–September) chlorophyll-*a* concentrations (0–2.4 m depth) for the period 2001–2006 and reference conditions for the respective areas. The red line indicates the target EQR of 0.67.

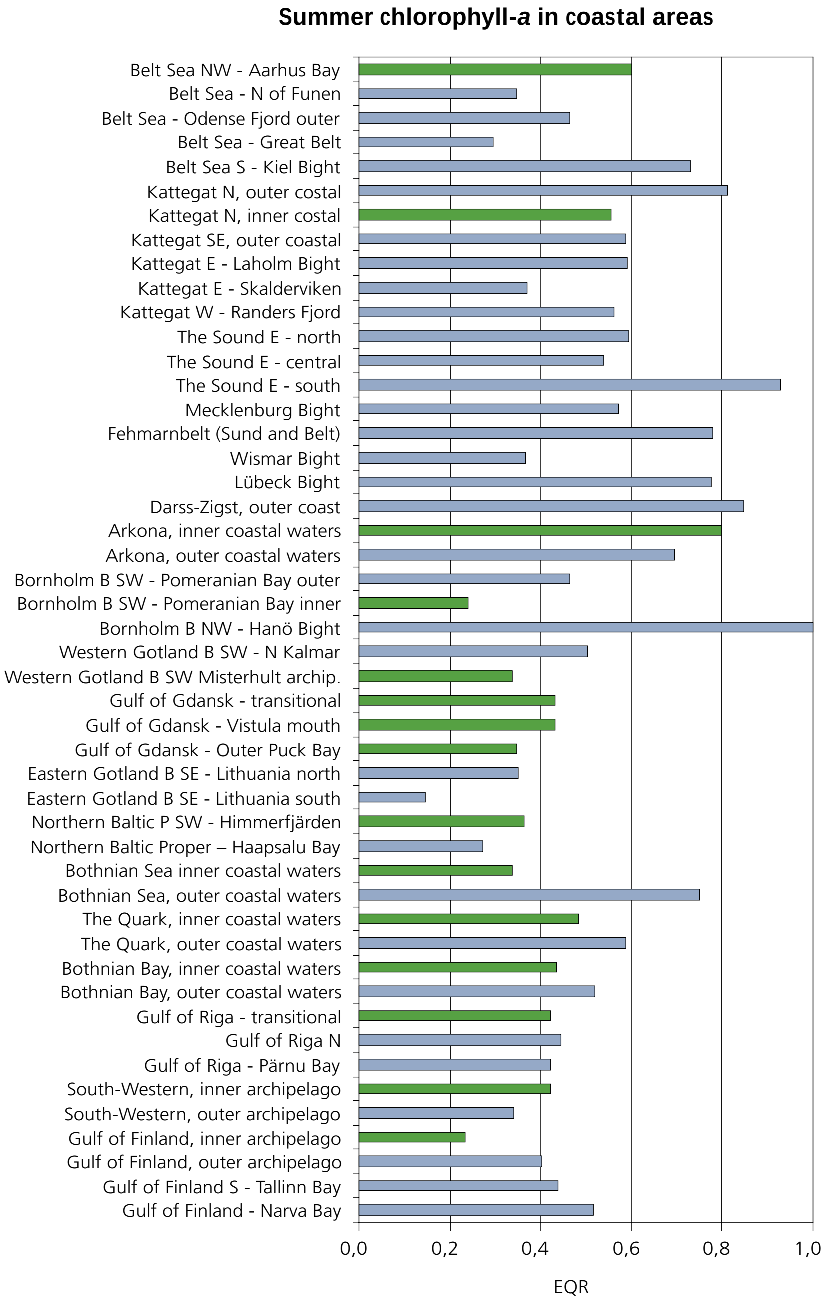


Figure 2.12 Chlorophyll status in Baltic Sea coastal areas expressed as Ecological Quality Ratio (EQR) values. The EQR values are based on the average summer (June–September) chlorophyll-*a* concentrations for the period 2001–2006 and reference conditions for the respective areas. The green colour denotes inner coastal areas and the blue colour outer coastal areas.

ern coastal waters, the EQR value was 0.35, but lower in the Lithuanian southern coastal waters (0.15). In the Gulf of Riga transitional and northern coastal waters and Pärnu Bay, the chlorophyll status values were very similar 0.42–0.44. In the Western Gotland Basin, the EQR value was 0.50 in the northern Kalmar Sound, but lower in the Misterhult Archipelago (0.34).

In the southwestern Gulf of Finland, the chlorophyll-based status in the inner and outer archipelagos (0.42 and 0.34, respectively) was comparable to the status in the inner and outer archipelagos of the northern Gulf of Finland (0.23 and 0.40, respectively). The EQR values in the Northern Baltic Proper also reflected the lowered status in the Himmerfjärden (0.36) and Haapsalu Bay (0.27).

Of the northern coastal waters of the Baltic Sea, the outer coastal waters of the Bothnian Sea are closest to reference conditions (EQR = 0.75). Typically, the EQR values were lower in the inner coastal waters than in the outer coastal waters. This was also notable in the coastal waters of the Bothnian Sea (inner 0.34, outer 0.75), the Quark (0.48, 0.59), and the Bothnian Bay (0.43, 0.52).

Chlorophyll-*a* concentrations derived from satellite remote sensing showed a large spatial variability in the Baltic Sea during the period 1998–2005 which might be related to the variability of the meteorological conditions in the basins and their catchments (Schrimpf & Djavidnia 2006). The July–August chlorophyll-*a* concentration was 2.3 µg l^{-1}, as averaged for the whole Baltic Sea area in 1998–2005. In the sub-basins, the chlorophyll concentrations can deviate significantly from an overall average. The high (>4 µg l^{-1}) chlorophyll-*a* concentrations were recorded in July–August in the Gulf of Finland and low (<2.5 µg l^{-1}) concentrations in the Gulf of Bothnia (Schrimpf & Djavidnia 2006). In the Baltic Proper, the chlorophyll-*a* concentrations showed high temporal and spatial variation in 1998–2005. According to the satellite images, the average values of chlorophyll-*a* in summer were lowest (0.5–2.0 µg l^{-1}) in the Kattegat and the Belt Sea.

The spring bloom of phytoplankton is a normal phenomenon in the Baltic Sea sub-basins. The spring bloom intensity index (Fleming & Kaitala 2006a), based on high-frequency monitoring data of chlorophyll-*a* in 2001–2006, has varied between 0 and 307 in the Arkona Basin, 220 and 522 in the Northern Baltic Proper, and 443 and 1060 in the Gulf of Finland (Fleming & Kaitala 2006a, 2006b). The spring bloom intensity has been calculated since 1992, and the index values from the period

2001–2006 are comparable to earlier year's results. No long-term trends were detectable in the spring bloom intensity index for the period 1992–2006 (Fleming & Kaitala 2006b). In the Gulf of Finland, this period coincides with decreased external loading of N (Pitkänen et al. 2007a), and decreased spring bloom maxima (Raateoja et al. 2005). A tendency to earlier spring blooms has been found in the Arkona Sea, the Eastern Gotland Basin, and the Kattegat (Wasmund & Siegel 2008).

2.3.2 Temporal trends

This section discusses temporal trends of chlorophyll-*a* in open sea areas of the Baltic Sea sub-basins based on an analysis of COMBINE data (0–2.4 m depth) and recent literature. In addition, the state and trends for certain other phytoplankton indicators are presented.

In the Kattegat and the Arkona Basin, the chlorophyll-*a* status increased from the 1970s to the 1990s in relation to reference conditions, but subsequently the status has again increased (**Fig. 2.13**). In the Bornholm Basin and the Eastern Gotland Basin, the chlorophyll status improved from the 1980s until the 1990s; since then the status in the Bornholm Basin has remained largely the same, whereas in the Eastern Gotland Basin the status has started to decrease again. The Western Gotland Basin, the Gulf of Riga and the Northern Baltic Proper show parallel temporal trends: the chlorophyll-*a* status first increased from the 1980s until the 1990s, then showed a decreasing trend in the 1990s that was again followed by an increase in the 2000s. In the Bothnian Sea, the Bothnian Bay and the Gulf of Finland, the chlorophyll status showed a deterioration from the 1970s/1980s until the 2000s (**Fig. 2.13**); however, in the Bothnian Bay and the Northern Baltic Proper the status has improved since 2004.

The temporal trends in the chlorophyll status presented in **Fig. 2.13** mostly agree with recent literature. Fleming-Lehtinen et al. (2008) analysed the changes in the chlorophyll-*a* concentrations (< 2 m) in June–September in the open northern Baltic Sea from 1972 to 2006. Their results show an increase of more than 150% in the surface chlorophyll-*a* concentrations in the Northern Baltic Proper (from 2.0 µg l^{-1} to 5.2 µg l^{-1}) and the Gulf of Finland (2.7 µg l^{-1} to 6.6 µg l^{-1}) from the 1970s to the present. An increase in chlorophyll-*a* of a similar magnitude (from 1.1 µg l^{-1} to 3.2 µg l^{-1}) was also observed in the Bothnian Sea from the late 1970s until the late 1990s, but after that the concentrations decreased to the present (2004–2006) level of 2.4 µg l^{-1} (Fleming-Lehtinen et al. 2008). The surface chlorophyll-*a* concentrations in the Bothnian Bay have remained at the same level (1.7–1.8 µg l^{-1}) during the past 30 years.

The increase in chlorophyll-*a* in the open areas of the Baltic Sea sub-basins was also confirmed by an analysis of June–September data from the period 1992–2006 (Jaanus et al. 2007). An increase in summer chlorophyll-*a* (0–10 m) was not observed in the Northern Baltic Proper or the Bothnian Bay. Linear regression of chlorophyll-*a* data (0–10 m depth) from 1979 to 2006 showed significant trends in spring data in the Mecklenburg Bight (decreasing), Arkona Basin (increasing), Bornholm Basin (increasing), and Eastern Gotland Basin (increasing), but no trends in summer data (Wasmund & Siegel 2008). In the Gdańsk Deep and the southeastern Gotland Basin, no trends could be observed in annual chlorophyll-*a* from the 1980s to 2005, although the summer chlorophyll-*a* values indicated some positive tendency (Łysiak-Pastuszak & Piątkowska, *unpublished data*). A recent analysis using different methodology (the pooled COMBINE open and coastal water data, including all seasons and stations from the Baltic Proper) showed a very slowly decreasing trend for median chlorophyll-*a* in the Baltic Proper from 1974 until 2005 (Håkansson & Lindgren 2008).

Phytoplankton primary production (PP) responds rapidly to changes in nutrient availability and, therefore, it has been suggested to be used in monitoring coastal zone eutrophication in the Baltic Sea (Andersen et al. 2006; HELCOM 2006). Long-term data sets of PP from the Baltic Sea are rare. The PP data from the past 20–50 years show large spatial and temporal variation in the Kattegat and the Belt Sea (Rydberg et al. 2006). In recent years, the daily PP has exhibited two distinct maxima, one in March and another between July and September, that were not recorded in the 1950s and 1960s. The results also indicate that annual PP has clearly increased since the 1950s, but this increase occurred before 1980 (Rydberg et al. 2006). As compared to the modelled reference

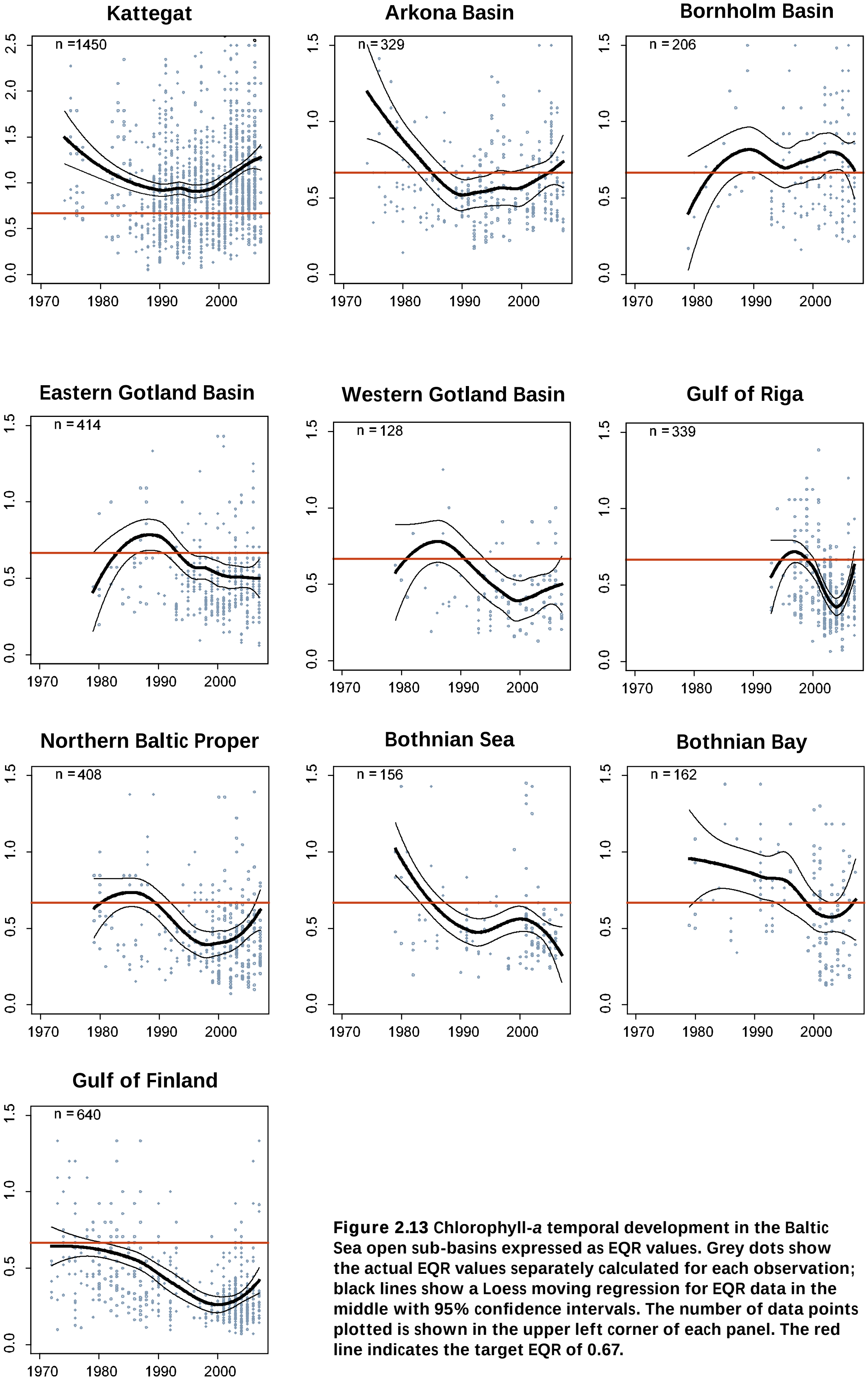


Figure 2.13 Chlorophyll-*a* temporal development in the Baltic Sea open sub-basins expressed as EQR values. Grey dots show the actual EQR values separately calculated for each observation; black lines show a Loess moving regression for EQR data in the middle with 95% confidence intervals. The number of data points plotted is shown in the upper left corner of each panel. The red line indicates the target EQR of 0.67.

Figure 2.14 **A bloom of cyanobacteria in the open sea (panel A) and sampling of a bloom (panel B).**

conditions for primary production (HELCOM 2009), since the 1980s the annual primary production in the southeastern Kattegat and the Sound (Rydberg et al. 2006) has been on average higher by 50% than the proposed reference conditions.

Shifts in the Baltic Sea phytoplankton communities are detectable in the long-term data sets. Diatoms and dinoflagellates dominate the spring bloom in the Baltic Sea (HELCOM 2002). The results suggest that dinoflagellates have become more abundant in spring in many parts of the Baltic Sea (Rahm et al. 1996; Kuparinen & Tuominen 2001; Wasmund & Uhlig 2003; but see Wasmund et al. 2008). The shift from diatoms to dinoflagellates in the spring can influence the nutrient dynamics in the summer and organic matter load to the sediment as the diatoms usually sediment to the seabed at the end of the bloom, whereas the dinoflagellates are mostly remineralized in the upper water layers (Heiskanen 1998; Tamelander & Heiskanen 2004).

The reason for the increased importance of dinoflagellates is not clear, but it may be linked to changes in climate conditions and stratification patterns (Heiskanen 1998; Wasmund et al. 1998; Spilling 2007; Toming & Jaanus 2007). In coastal regions, eutrophication often leads to lower availability of silica (Si) resulting in a decrease in the intensity of the diatom blooms and changes in the species in the diatom community (e.g., Olli et al. 2008). Once Si is depleted, high levels of inorganic P may promote dinoflagellates, in particular when the N:P ratio is low (Howarth & Marino 2006). Based on an analysis of sediment core diatom data from four sub-basins, strong Si depletion has been evident in the Gulf of Riga, but not in the Kattegat, the Baltic Proper, or the Gulf of Finland (Olli et al. 2008).

Cyanobacteria dominate in most sub-basins of the Baltic Sea during the summer period (HELCOM 2002; Jaanus et al. 2007). Satellite images show that in the period 2001–2006 cyanobacterial bloom days were the highest in 2005 when blooms were observed in the whole Baltic Proper area and the Gulf of Finland (Hansson 2007). The bloom duration and extent in 2005 was very similar to the situation in 2003. Cyanobacterial blooms were common in the northern Baltic Proper and the Gulf of Finland in 2002, while in 2006 they were mostly observed in the south (Hansson 2007). The frequency and magnitude of the algal blooms, as derived from satellite images and averaged over the whole Baltic Sea area, have varied during 1997–2006 without a trend (Schrimpf & Djavidnia 2007; Hansson 2007).

Cyanobacterial biomass has been lower in the 2000s than in the 1980s–1990s in the Gulf of Riga, Eastern Gotland Basin, and Arkona Basin (Jaanus et al. 2007). The late summer biomass of cyanobacteria has been suggested to have increased in the northern Baltic Sea since the late 1970s (Suikkanen et al. 2007). The cyanobacterium *Aphanizomenon flos-aquae* (**Fig. 2.14**), which is one of the dominating cyanobacteria in the Baltic Sea, has been suggested to increase in the uppermost 10 m in the Gulf of Finland from 1968 to 2004 (Fleming-Lehtinen 2007; Suikkanen et al. 2007). In the northern Baltic Proper (Askö), an analysis of data from 1990–2006 shows that *Aphanizomenon* spp. biomass has increased in 2003–2006 (Hajdu et al. 2007).

The cyanobacterium *Nodularia spumigena*, which forms toxic blooms in the Baltic Sea, has also been reported to increase in the late summer in the open northern Baltic Sea (Suikkanen et al. 2007). The cyanobacteria bloom index, which integrates the rank abundance of *Aphanizomenon* and *Nodularia* during the whole growth season in the Baltic Sea, indicates that the bloom intensity of *Nodularia* has remained unaltered during the 2000s (Kaitala & Hällfors 2007). In the coastal waters of the Gulf of Gdańsk, intensive blooms of *N. spumigena* have been recorded frequently since 1994 (Mazur-Marzek et al. 2006). The largest *Nodularia* blooms have occurred in 1994, 2001, 2003, and 2004.

2.4 Water transparency

Water transparency is a measure of the clarity of the water. It indicates the attenuation of light penetrating into the water column and is governed by the absorption and scattering properties of the water. Scattering and absorption of light is dependent on the amount of particulate matter and dissolved substances in the water. The material in the water is typically living or dead organic particles (e.g., phytoplankton), small inorganic particles, or a combination of both, and dissolved coloured substances, e.g., humic substances. **Fig. 2.15** shows a conceptual model of factors governing water transparency in the Baltic Sea.

Secchi depth, a measurement of water transparency, can be used as an indicator of eutrophication because increasing phytoplankton biomass is likely to have a large influence on water transparency during summer (in this assessment, defined as from June to September). Changes in water transparency can thus be used to provide an indication of changes in summer phytoplankton biomass.

Water transparency is most often measured by using a white Secchi disc of about 30 cm in diameter, which is lowered through the water column (**Fig. 2.16**) and the disappearance depth is determined by eye (e.g., Preisendorfer 1986). The method is very simple and inexpensive, and the white Secchi disc is one of the few early hydrological measuring devices still in use. In the Baltic Sea, observations have been made from the end of the nineteenth century to the present (Sandén & Håkansson 1996; Aarup 2002; Fleming-Lehtinen et al. 2007). Thus, available Secchi depth records provide unique first-hand information on environmental changes in the Baltic Sea, starting from a time when it was in a near-pristine state.

In the assessment of eutrophication, water transparency reflects direct effects of eutrophication,

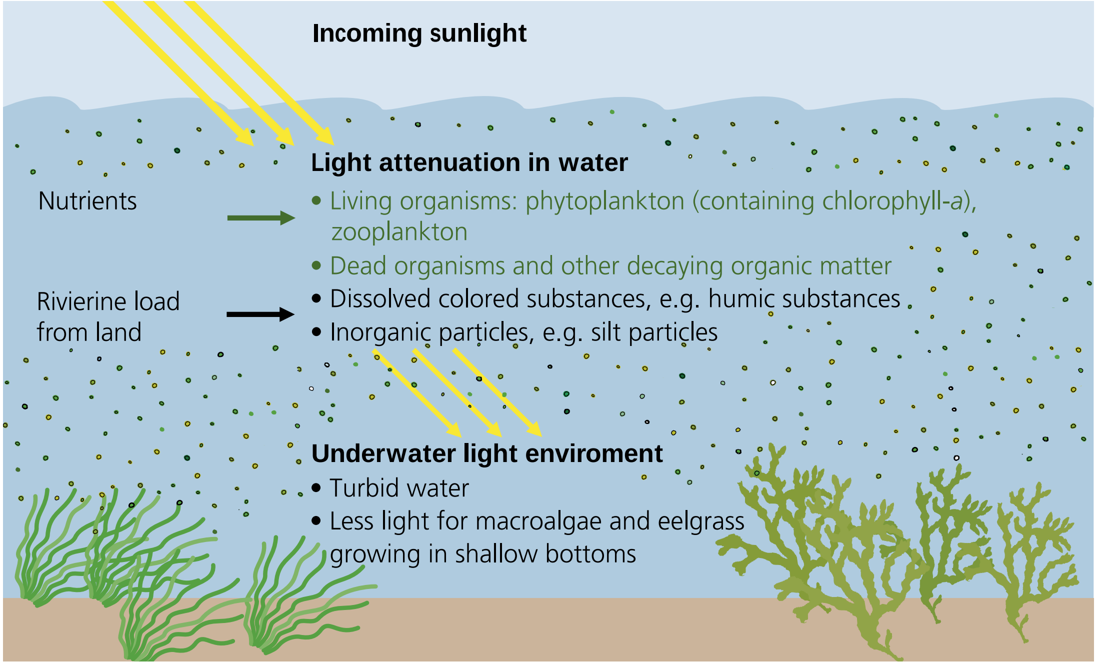


Figure 2.15 A conceptual model of factors affecting water transparency in the Baltic Sea. Eutrophication-driven factors are highlighted in green.

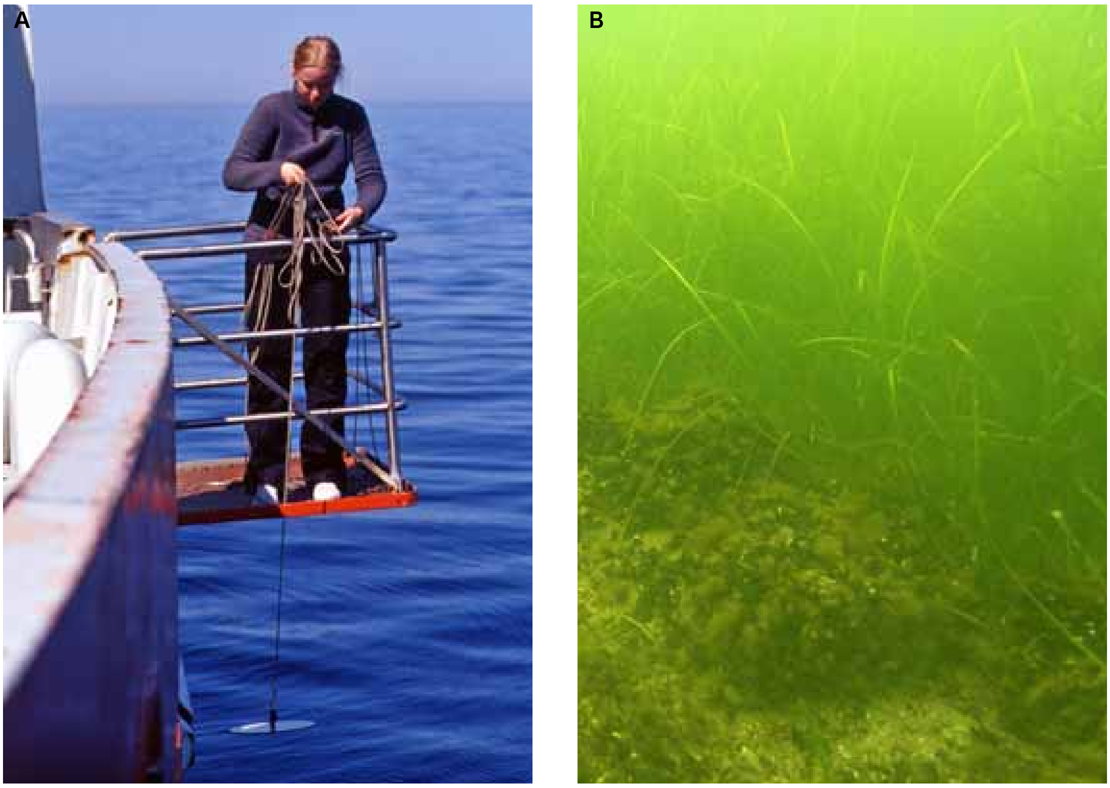


Figure 2.16 **Secchi disk used for measurements of water transparency (panel A) and low water transparency (panel B).**

i.e. an increase in phytoplankton biomass. Water transparency regulates the amount of light in underwater environments and thus is a central parameter for all aquatic photosynthesis. In coastal areas, the depth limit of aquatic macrophytes decreases as water transparency decreases, narrowing the width of important coastal habitats.

In the HELCOM system of Ecological Objectives (EcoOs), water transparency is directly linked to the EcoO 'clear water' and is one of the HELCOM Baltic Sea Action Plan (BSAP) indicators for eutrophication. In addition, as an integrative parameter, water transparency can be used indirectly to illustrate the changes in phytoplankton biomass related to EcoO 'natural level of algal blooms'.

The significance of phytoplankton biomass for changes in Secchi depth may vary according to sea area, time period or distance from the shore. In areas with relatively low algal production and a large amount of other particulate or dissolved organic matter, a correlation between algal biomass and water transparency can be weaker than in areas where algal biomass is high and the water non-coloured.

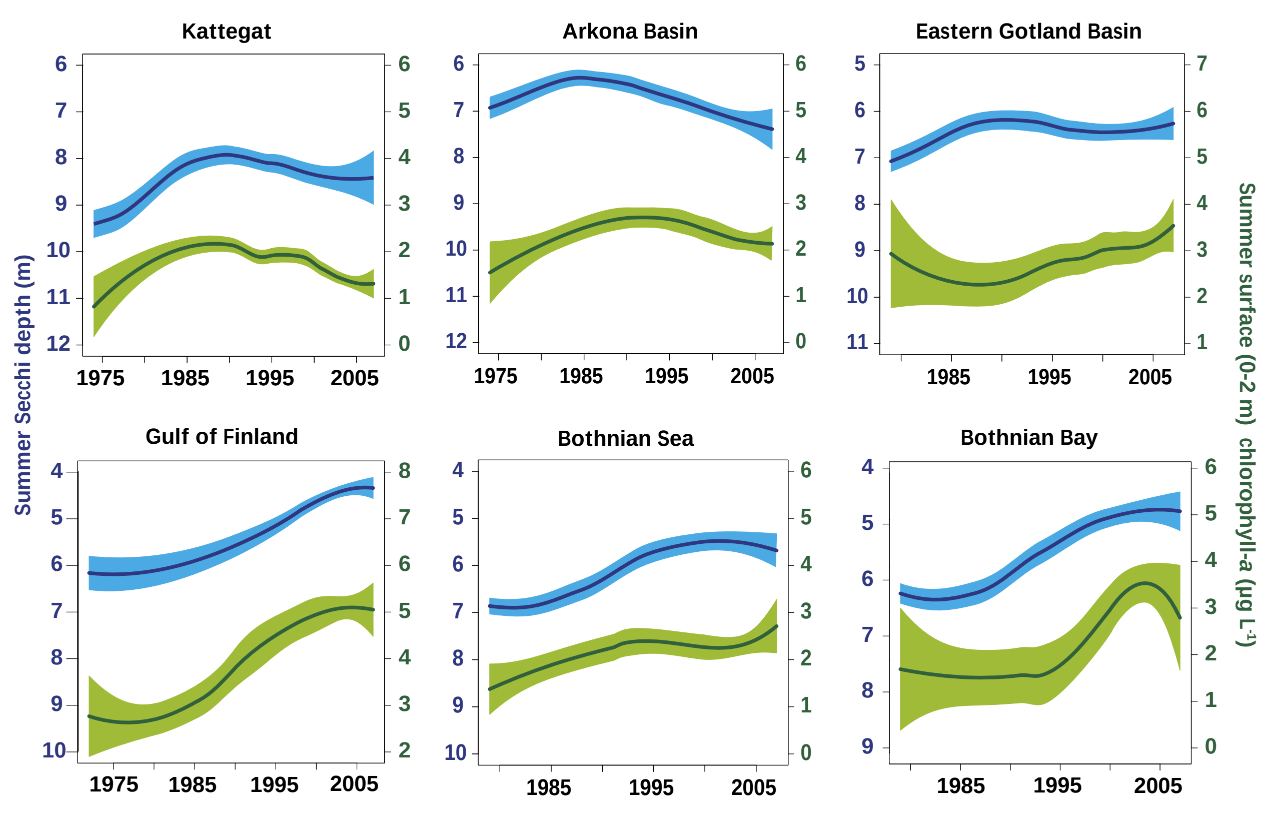
An analysis of the share of light attenuation by phytoplankton in northern Baltic Sea open basins (HELCOM 2009) clearly shows that phytoplankton biomass is an important factor for water transparency and changes in phytoplankton biomass are reflected in Secchi depth, but other factors, e.g. changes in organic matter load can also be important especially in the northernmost sub-basins. This is shown, for example, by a decreasing Secchi depth in the Bothnian Bay with no parallel increase in chlorophyll-*a* concentration (Fleming-Lehtinen et al. 2008). As the contribution of chlorophyll-*a* to water transparency seems to be rather consistent in all open-sea areas of the Baltic Sea, direct effects of eutrophication should also be reflected in a similar manner to water transparency in different open-sea basins.

A station-based approach was applied in the coastal water transparency assessment. Reference conditions and acceptable deviations are defined using different methods in the participating Contracting Parties. Methods used for reference setting include direct use of historical data, statistical modelling, empirical relationships between chlorophyll-*a* or total nitrogen concentrations and

BOX 3: Correlations between water transparency and phytoplankton biomass

Phytoplankton is the most important source of new organic matter in aquatic ecosystems, including the Baltic Sea. Phytoplankton fuels the food webs of the ecosystem, whose largest organisms, especially fish, are also efficiently exploited by human populations around the sea. The amount of phytoplankton production largely determines the total productivity of the entire ecosystem. When phytoplankton proliferates in water, it requires light for its growth and energy production. At the same time, phytoplankton cells, alive, dead, or decaying, also add turbidity and colouration to the water. Other organisms, such as certain zooplankton that are dependent on phytoplankton for their growth, also increase and add their share to water turbidity. Turbidity and colouration of water decrease water transparency as they attenuate light entering the water.

Phytoplankton standing biomass (living phytoplankton in water) is often measured by means of chlorophyll-*a*, a main photosynthetic pigment of many phytoplankton groups. The illustration in panel A, below, presents temporal development of water transparency (Secchi depth) and phytoplankton biomass, showing parallel trends in many areas during the past 25–30 years. Importantly, decreasing phytoplankton biomass, e.g. in the Kattegat and Arkona Basin, has led to an improvement in water transparency, although changes in water transparency may be somewhat slower than changes in phytoplankton biomass. In some areas, such as the Gulf of Bothnia, water colouration mainly by humic substances from the catchment greatly affects the water transparency and the relationship between phytoplankton biomass and water transparency is not as clear. It should also be noted that significant changes in water transparency took place in some areas already before the time that chlorophyll-*a* measurements commenced in the early 1970s (see below).



Panel A: Temporal development of water transparency (Secchi depth, shown in blue) and phytoplankton biomass (expressed as chlorophyll-*a*, shown in green) in six selected Baltic Sea sub-basins. The lines indicate Loess moving regressions for the entire data set and shading shows the 95% confidence intervals for the regression line.

Secchi depth, and the use of historical depth limits of aquatic macrophytes. The acceptable deviation is in most cases defined to correspond to the WFD classification or the previously agreed normative upper limit of an acceptable deviation in the order of minus 25% from reference conditions (HELCOM 2006, 2009).

In the open-sea assessment, an area-based approach was applied (Fleming-Lehtinen 2007). All available summer (June–September) observations were pooled together to enable a reliable assessment on a sub-basin scale. Reference conditions were defined using the five-year period, with the oldest data from the beginning of 20th century. Data used in the open-sea assessment comprised all available data from the ICES database as well as additional data from several Contracting Parties (Finland, Latvia, Lithuania, Poland, and Sweden). The status for the years 2001–2006 was defined as the mean of all summer observations for the assessment period.

Temporal trends in water transparency were assessed for open-sea areas only, based on data included in Fleming-Lehtinen et al. (2007) and assessment data. Secchi depth data are available since 1903, but temporal trends are considered mainly from the 1960s to the present owing to adequate data coverage for that period.

2.4.1 Status 2001-2006

Water transparency status varied notably among the sub-basins of the Baltic Sea (**Fig. 2.17**). The status was best in the Arkona Basin and the Kattegat, acceptable in the Bornholm, Eastern and Western Gotland Basins and the Gulf of Riga, and significantly lower in the Northern Baltic Proper, Gulf of Finland and Gulf of Bothnia. Water transparency status was generally lower in inner coastal and transitional waters and increased in outer archipelagos and the open sea, but it generally followed the same large-scale geographical pattern in coastal and open-sea areas. The assessment here has been conducted on a sub-basin scale, although water transparency also varied strongly within sub-basins, as shown in **Fig. 2.17**. In particular, central parts of the Bothnian Bay, Bothnian Sea, and the Northern Baltic Proper have higher a status than open-sea areas near coast. In the Western and Eastern Gotland Basins, the water transparency status was better in the southern than in the northern parts.

Open sea

In the open-sea assessment, ten open-sea sub-basins were included. Water transparency reference conditions varied between 4.0 m and 10.5 m, with a median of 8.0 m for all sub-basins. The actual status in 2001–2006 varied between 3.0 m and 8.5 m, with a median value of 6.0 m.

The water transparency status in open-sea areas expressed as Ecological Quality Ratio varied notably in the different sub-basins of the Baltic Sea. The status varied from 0.75 to 0.94 expressed as EQR values for the southern and central sub-basins assessed, indicating a 6% to 25% decrease from near-pristine reference conditions. Sub-basins north of the Northern Baltic Proper exhibited a significantly lower status, with EQR values ranging from 0.50 to 0.61, representing a reduction of 39% to 50% in water transparency compared to reference conditions. The mean EQR value for all open sub-basins assessed was 0.73. The water transparency status for open sea areas is presented in detail below.

In the Kattegat, water transparency status in open-sea areas clearly exceeded the mean status (EQR 0.81). In the Arkona Basin, water transparency status was the highest of all open sub-basins, with an EQR value of 0.94. In the Bornholm Basin and Eastern Gotland Basin, the status was nearly equal to that of the Kattegat (0.78 and 0.81, respectively). In the Western Gotland Basin and Gulf of Riga, the status was comparable to or only slightly lower than in the Eastern Gotland Basin, with EQR values of 0.75 for both areas.

The Northern Baltic Proper and Gulf of Finland represent remarkably low status compared to reference conditions, with EQR values of 0.61 in the open Northern Baltic Proper and 0.50 in the Gulf of Finland. The open-sea area water transparency status in the Bothnian Sea was 0.61 and in the Bothnian Bay 0.56 expressed as EQR. However, in these two areas the contribution of water colour changes (related, for example, to changes in land use) to water transparency may be greater than in other sub-basins; especially in the Bothnian Bay, the relative importance of phytoplankton to water trans-

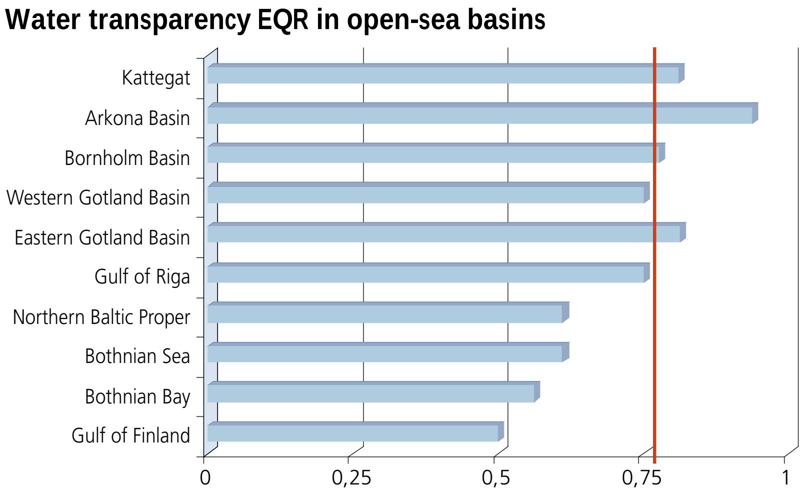


Figure 2.17 Water transparency status in Baltic Sea open sub-basins expressed as ecological quality ratios. The red line indicates the target EQR of 0.75.

parency has been shown to be lower (see temporal trends section below and HELCOM 2009).

Coastal waters

The water transparency status is typically lower in the inner coastal and transitional waters compared to the open-sea areas owing to higher phytoplankton concentrations and higher particle and/or humic content originating from shallow sediments or river discharge. However, in the Gulf of Finland, southern coastal waters have a higher water transparency status than open-sea areas, which may be related to water circulation patterns in the Gulf of Finland or the whole-area approach taken in the open-sea assessment, levelling off intra-basin spatial variations (see **Fig. 2.17**). The water transparency status in coastal waters generally followed the geographical pattern observed in open-sea areas: coastal waters in the Kattegat, Arkona and Bornholm Basins as well as the Eastern and Western Gotland Basin had somewhat higher status, whereas in the Northern Baltic Proper, Gulf of Finland and Gulf of Bothnia, where the status of open waters is low, coastal waters also had lower status. In the Gulf of Riga, coastal waters had remarkably lower status in comparison to open waters.

Reference values for the 51 coastal areas included in the assessment range from 4.0 m to 13.7 m, reflecting the highly diverse nature of the coastal areas along the geographical expanse of the Baltic Sea. The median value for coastal reference conditions is 7.50 m, which is close to the open-sea median value of 8.0 m. The status values varied from 1.8 m to 9.10 m, with a median value of 4.30 m, resulting in a mean EQR of 0.58 for all coastal areas corresponding to an average 42% reduction in water transparency from near-pristine reference conditions. All EQR values for the coastal areas assessed are presented in **Fig. 2.18**, and the status of water transparency in 2001-2006 is mapped in **Fig. 2.19**.

Kattegat and the Sound

The water transparency status in Kattegat coastal waters is clearly lower than in the open Kattegat. EQR values were 0.66 in both the eastern central inner coastal waters and coastal Laholm Bay; another coastal bay (Skälder Bay) had a lower EQR value comparable to outer Randers Fjord in the western Kattegat (EQR for outer parts 0.56 and inner parts 0.44). In the northern Sound (EQR 0.78) and central (western) open waters (EQR 0.71) water transparency status was close to that in the open Kattegat, whereas the central eastern (EQR 0.58) and southern Sound (EQR 0.59) showed a lower status comparable to the coastal Kattegat.

Belt Sea, Arkona Basin

Water transparency status in the Belt Sea sites was rather homogeneous. The status expressed as EQR varied between 0.59 and 0.70. Coastal Geltinger Bay had the highest status, 0.70, close to that in Lübeck Bight (0.67), southwestern and northwestern Kiel Bight (EQRs 0.64 and 0.67,

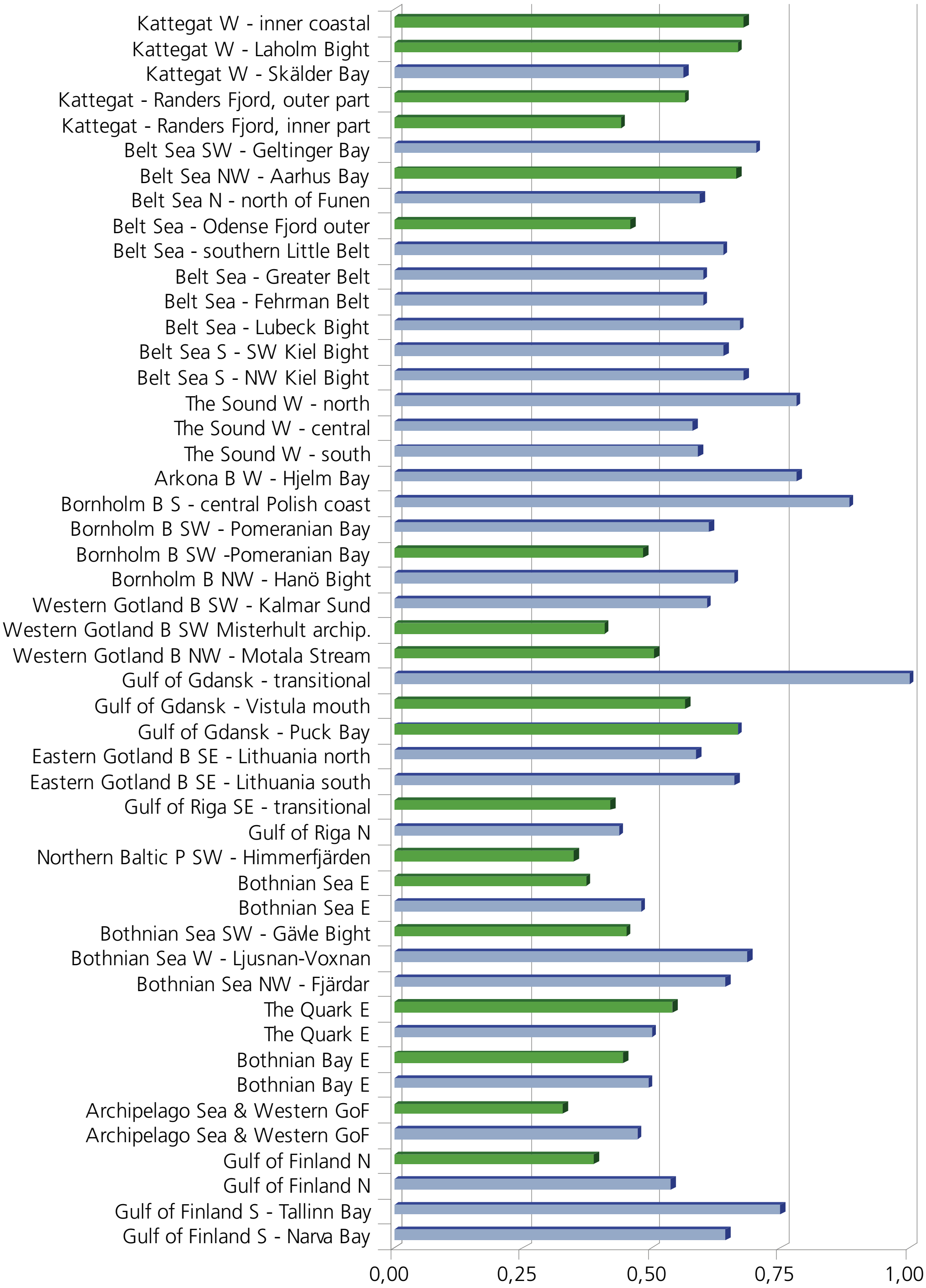


Figure 2.18 Water transparency status in Baltic Sea coastal areas expressed as Ecological Quality Ratios. The green colour denotes inner coastal sites and the blue colour outer coastal sites.

respectively), and Aarhus Bay (0.66). Fehrman Belt (0.60), greater and southern Little Belt (0.60 and 0.64, respectively), as well as waters north of Funen (0.59) had almost equal status. In Odense Fjord outer waters, the status was clearly lower (0.46). In the Arkona Basin, there was only one coastal site, Hjelm Bay in the western Arkona Basin, with the relatively high EQR value of 0.78, which is consistent with the high status of the open Arkona Basin (see above).

Bornholm Basin

Water transparency status in the open Pomeranian Bay (EQR 0.61), transitional waters at the mouth of the river Oder in Pomeranian Bay (EQR 0.48) as well as in coastal Hanö Bight (EQR 0.67), on the southeast coast of Sweden, was rather similar to other coastal areas in the southern sub-basins. In contrast, the central Polish coast with a more open nature showed high water transparency status (EQR value 0.88), comparable, for example, to the high status of open Gulf of Gdansk waters.

Eastern Gotland Basin, Gulf of Gdansk, Gulf of Riga

In internal Gulf of Gdansk open waters, the status exceeded the reference conditions (EQR value 1.03), whereas the status was lower in the river Vistula mouth (EQR 0.57) and Outer Puck Bay in the Gulf of Gdansk (EQR 0.67). This difference may be related to the overall improvement of water transparency in the open Eastern Gotland basin (see temporal trends section, below) affecting open Gulf of Gdansk waters. In Lithuanian southern open sandy coastal waters, the water transparency status (EQR 0.66) was rather similar to that in many other coastal areas in southern sub-basins but it was higher (0.86) in the northern rocky open coastal water types, even exceeding the open-sea status for the Eastern Gotland Basin (see above). In Gulf of Riga transitional and northern coastal waters, the transparency status was rather low (0.42 and 0.44, respectively) and comparable to that in the northern and western parts of the Gulf of Finland (see below).

Western Gotland Basin

In the outer coastal waters of the Kalmar Sound between Öland Island and mainland Sweden, the water transparency status (EQR 0.61) was close to that in the open Western Gotland Basin, but it was lower in inner coastal areas in Misterhult Archipelago, with an EQR value of 0.41, and Motala stream (EQR 0.49).

Northern Baltic Proper, Gulf of Finland

In the western Gulf of Finland, the Archipelago Sea and the Åland islands sheltered inner archipelagos, where water circulation is weak, the water transparency was significantly reduced compared to reference conditions, with the lowest EQR value of all coastal sites assessed (0.33). Outer, open waters in these archipelagos exhibited somewhat higher status (EQR 0.47). On the western coast of the Northern Baltic Proper, the status of Himmerfjärden (EQR 0.35) was comparable to that in the inner northeastern Baltic Sea coastal waters.

On the northern coast of the Gulf of Finland, water transparency status in sheltered, shallow inner archipelago areas with reduced water circulation was low in comparison to reference conditions (EQR 0.39), whereas in the open, outer archipelago the status was higher (EQR 0.54). On the more open southern coast of the Gulf of Finland, water transparency status was higher than in open archipelagos on the northern coast, with EQR values of 0.64 in Narva Bay in the eastern Gulf of Finland and 0.75 in Tallinn Bay in the central Gulf.

Bothnian Sea, the Quark area and Bothnian Bay

On the eastern side of the Bothnian Sea, the relatively open inner coastal waters, with a complex shoreline and little islands and island groups, the water transparency status was comparable to that in the northern Gulf of Finland inner coastal waters (EQR 0.37), despite the more exposed nature of the coast. Outer coastal waters of the Bothnian Sea, with only little islands and skerries, had a somewhat higher status (EQR 0.48). On the western coast of the Bothnian Sea, two areas had a higher water transparency status (Fjärdar, EQR 0.65) and Ljusnan-Voxnan (EQR 0.69) than on the eastern coast, whereas Gävle Bight showed a lower status (EQR 0.44), comparable to inner coastal waters in the eastern Bothnian Sea.

In the eastern Quark area, water transparency status was comparable to that in the outer archipelagos of Finnish coastal waters, with EQR values of 0.50 and 0.54 in sheltered inner and more open outer archipelagos, respectively. Water transparency status was 0.44 in the eastern inner and 0.49 in eastern outer Bothnian Bay coastal waters, both of which are relatively open. However, water colouration owing to high humic substance content has a larger influence on water transparency in the Bothnian Bay.

2.4.2 Temporal trends

A decrease in summer water transparency compared to reference conditions has been observed in the open sea in all Baltic Sea sub-regions over the past one hundred years (**Fig. 2.20**). A pronounced decrease over the past 25 years has occurred in the Gulf of Riga and the Northern Baltic Proper, and the Gulf of Finland and Gulf of Bothnia. In the Kattegat, Arkona Basin, Bornholm Basin, and Eastern Gotland Basin, the decreasing trend ceased during the past 15 to 20 years and since then the water transparency status has been improving.

There are three main groups of temporal trends in water transparency among the sub-basins. The first two groups are rather uniform, while the third group contains two areas with unique responses. 1) In the Kattegat, Arkona Basin and Bornholm Basin, a decreasing trend from the 1960s until the late 1980s or early 1990s prevailed before a trend reversal and improving water transparency status up to the present. 2) In the Northern Baltic Proper, Gulf of Finland, Bothnian Sea and Bothnian Bay, water transparency was already significantly lower in the 1960s and 1970s when modern measurements commenced. In all four northern sub-basins, but most clearly in the Gulf of Finland and the Bothnian Sea, a period with steady water transparency status lasted until the early 1980s when further deterioration of the water transparency status began and still continues. 3) In the Eastern Gotland Basin, water transparency decreased until the late 1980s and has since remained rather stable up to the present. In the Western Gotland Basin, water transparency increased from the 1960s to the mid-1980s, when the current downward trend started. In the Gulf of Riga, elevated water transparency status compared to reference conditions prevailed in the late 1950s, when modern measurements began, and water transparency has been steadily decreasing ever since.

Although the general decreasing trend in water transparency can be attributed to increased organism biomass and the degree of eutrophication in the respective sub-basins, the factors involved in the observed water transparency temporal trends are not entirely clear and need further study. In the Gulf of Riga, low reference conditions compared to other areas as well as a status exceeding the reference in the 1950s–1980s may reflect dynamic variation in the river discharge-influenced and naturally turbid waters in the Gulf. In the Bothnian Bay, water transparency had been decreasing already in the 1920s, which was not attributable to eutrophication changes in the area. Chlorophyll-*a* concentrations have also remained unchanged in the Bothnian Bay since the early 1980s (Fleming-Lehtinen et al. 2008) despite the remarkable decrease in water transparency status. Possible explanations include increasing CDOM (Coloured Dissolved Organic Matter, e.g. leached organic substances from the catchment due to changes in land use) or an increasing amount of particles other than phytoplankton during summer, e.g. heterotrophic organisms (see **Fig. 2.15**). It has been suggested that the Bothnian Bay is a net heterotrophic ecosystem (Sandberg et al. 2004), with dissolved carbon load from land being an important factor regulating the productivity of the system. The observed trend reversals in the southern sub-basins and the Gotland Basin can be linked to decreasing nitrogen concentrations (see **Chapter 2.1**), but they are also partly related to changes in large-scale hydrographic events during the observation period, e.g. the number of salt-water inflows and amount of river discharge (Mätthaus 2006; Meier 2007).

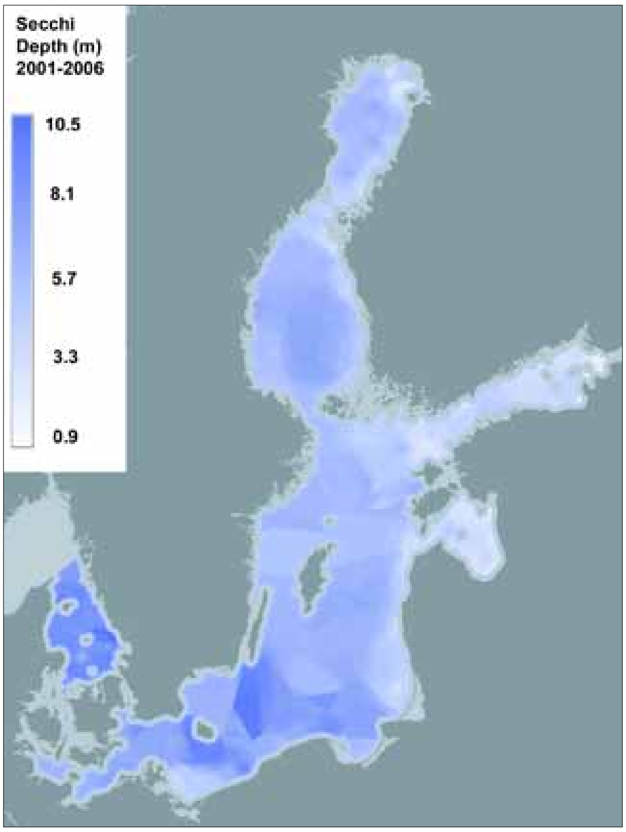


Figure 2.19 Map showing the water transparency status in 2001–2006 in the Baltic Sea area. Rectangular forms on the map are produced by a map interpolation method and do not represent boundaries in water transparency status.

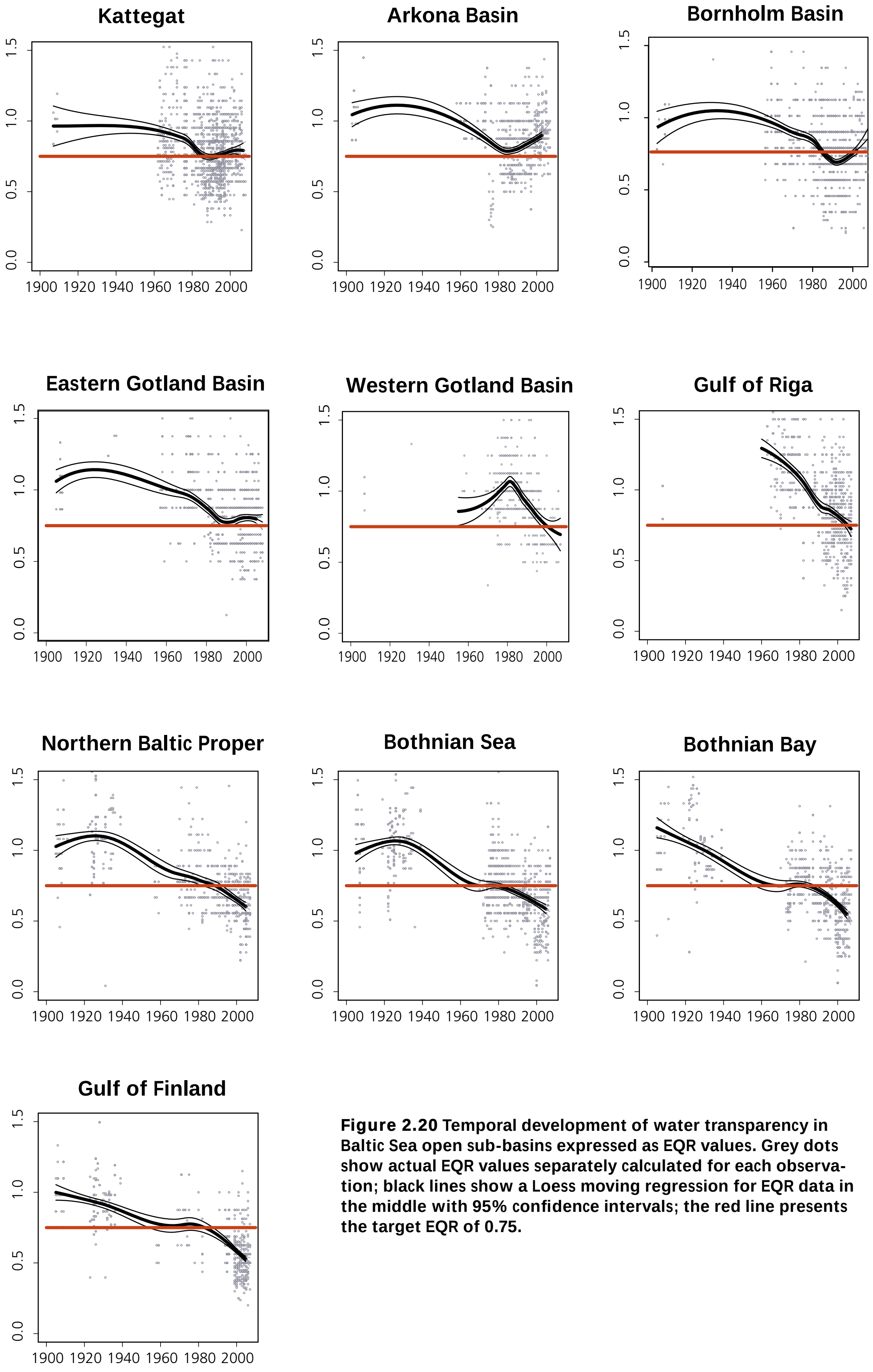


Figure 2.20 Temporal development of water transparency in Baltic Sea open sub-basins expressed as EQR values. Grey dots show actual EQR values separately calculated for each observation; black lines show a Loess moving regression for EQR data in the middle with 95% confidence intervals; the red line presents the target EQR of 0.75.

2.5 Submerged aquatic vegetation

Submerged aquatic vegetation (SAV) is an important component of coastal ecosystems in the Baltic Sea, where different geomorphological conditions create a large diversity of habitats suitable for underwater vegetation. Submerged aquatic vegetation is also an excellent indicator for ecosystem health as it plays a key role in the functioning of these ecosystems, channelling the energy and matter flows, and providing a habitat for numerous species of associated fauna and flora.

At present, 442 species of macroalgae are found in the Baltic Sea, including the Kattegat area (Nielsen et al. 1995). As is typical for most brackish water systems, the number of marine species decreases with the salinity. In general, the pattern of decline of the number of species belonging to the Bangiophyceae and Fucophyceae and the increase in the Chlorophyceae species is well described along the falling salinity gradient in the Baltic Sea (Nielsen et al. 1995). Salinity is the main environmental factor controlling the distribution of species on the Baltic Sea-wide scale, while exposure, substrate type and light availability determine the structure of vegetation communities on the local scale (Kautsky 1988; Martin 2000). Eutrophication is a strong modifying factor for many types of vegetation communities, changing competitive interactions and the structure of the trophic web and, in general, influencing the habitat quality by changes in light climate and sedimentation rates.

Historical records describing the species composition of benthic macro-vegetation in the Baltic Sea date centuries back, while georeferenced quantitative data on the distribution of different species in most cases only cover the past 30–40 years, since adequate sampling and observation techniques (e.g. SCUBA diving) became available.

Regular monitoring programmes aiming at describing the quantitative and qualitative characteristics of benthic vegetation in relation to anthropogenic impacts on the Baltic Sea environment have been established in almost all countries around the Baltic Sea. The HELCOM COMBINE programme includes phytobenthos as a main parameter and monitoring guidelines have been available since 1999. Submerged aquatic vegetation is one of the Ecological Quality Elements (QE) used in EU Water Framework Directive (WFD) assessment schemes, so monitoring of this QE has must carried out in all coastal water bodies. The drawback is that in most of the coastal countries these monitoring programmes have either limited time series (programmes established only in recent years) or cover limited geographical areas. Another important limitation of these programmes is the dissimilarity in methodology and monitored indicators. Thus far, no large-scale (Baltic Sea-wide) assessments have been produced based on phytobenthos monitoring data.

Functional relations between different abiotic parameters and SAV are well established. It is known that the depth distribution of macrophytes is largely determined by light (e.g. Duarte 1991; Nielsen et al. 2002a) and, therefore, also by parameters affecting the light climate. Increased nutrient concentrations stimulate the production of phytoplankton and epiphytes, which reduce water clarity and thereby reduce the depth penetration ability of macrophytes (Nielsen et al. 2002a, 2002b). Thus, the depth distribution of macrophyte species or communities should respond predictably to the level of eutrophication (**Fig. 2.21**).

Similarly, increased nutrient concentrations can affect the phytobenthic community structure and dynamics. Under conditions with elevated nutrient concentrations, some plant species may be favoured in competition for light and space and some grazer species may benefit from the increase of their preferred food source. This can result in irreversible changes in community structure (Kotta et al. 2000). Thus, the species composition and quantitative structure of the community not only can reflect the trophic conditions but also illustrate trends in the development of the eutrophication process.

There are many different SAV parameters that can be used as indicators of the state of the environment and eutrophication. Many of these are characterized by large natural variability and some of them are sensitive to changes in living conditions of phytobenthos on a local scale (i.e. site specific). For the Baltic Sea conditions, it is possible to define more than 30 different indicators for SAV, all of which have to some extent the ability to reflect the trophic status of the coastal water, whereas only some of them can be used on a larger scale (basin or Baltic Sea scale). The evaluation and selection of

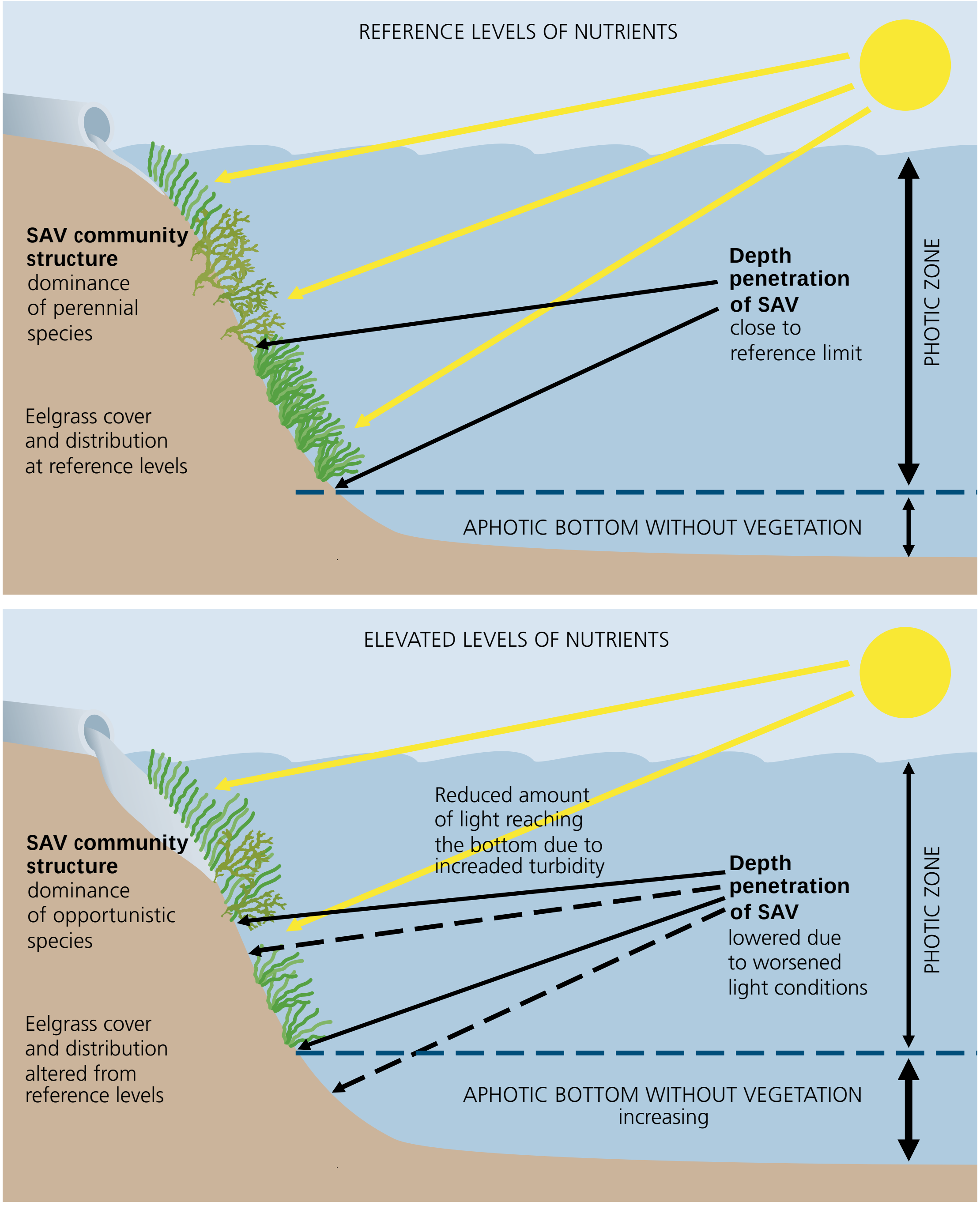


Figure 2.21 A conceptual model of factors affecting submerged aquatic vegetation in the Baltic Sea.

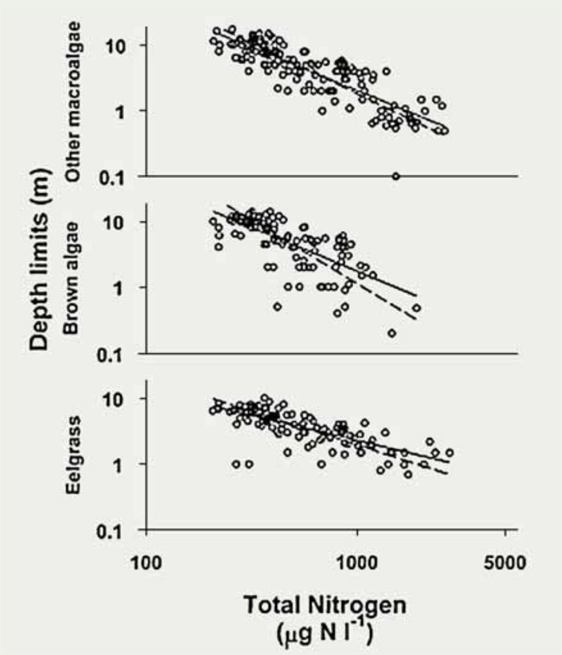
the indicators have been carried out both nationally, while preparing the assessment schemes for the purpose of the WFD, and internationally by different international projects.

Because this assessment has the aim of providing a comprehensive evaluation of eutrophication status on the scale of the entire Baltic Sea using well-established indicators, only one primary SAV indicator with the best-documented response to eutrophication and the best data coverage has been selected together with two supporting indicators to obtain the maximum geographical coverage. These are: (1) depth distribution of bladderwrack (*Fucus vesiculosus*) as the main indicator, and (2) distribution characteristics of eelgrass (*Zostera marina*), and (3) proportion of opportunistic species in the SAV community as the two supporting indicators.

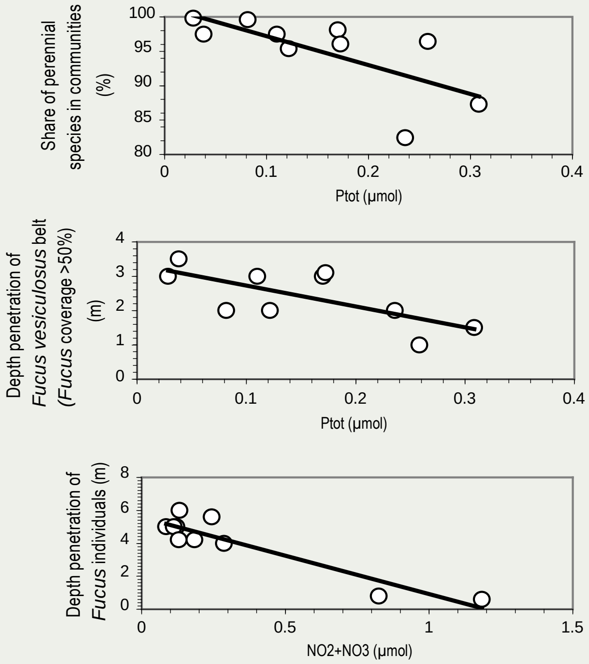
BOX 4: Correlations between submerged aquatic vegetation and nutrient enrichment

Submerged Aquatic Vegetation (SAV) reacts to changes in the trophic state of the environment by changes in community structure (elevated levels of nutrients favour filamentous annual species) and changes in depth distribution of species and communities (higher pelagic primary production lowers the water transparency and narrows the photic zone).

In order to establish correlations between different community parameters of submerged aquatic vegetation and nutrient concentrations, a large amount of high frequency monitoring data is required. These relationships are difficult to establish mainly owing to the different scales in the variability of these parameters. Nevertheless, several cases demonstrate that the direct influence of local or regional nutrient loadings and nutrient concentrations on abundance and distribution patterns of aquatic vegetation is possible to describe within existing monitoring efforts.



Panel A: Relationships between different SAV parameters and concentrations of total nitrogen in sea-water for Danish coastal areas (based on Sand-Jensen et al. 1994).



Panel B: Relationships between different SAV parameters and winter concentrations of nutrients for the southern Gulf of Finland (mean values for time period 1997–2007).

2.5.1 Current status

The status of SAV communities depends on a multitude of factors where eutrophication plays a key role especially in sheltered coastal areas. The use of selected, finely tuned indicators enables a filtering out of the influence of differences in environmental settings between the regions and highlights the signal caused by changes in factors related to eutrophication.

Depth limits of bladderwrack *(Fucus vesiculosus)*

Bladderwrack can be considered the most common habitat-forming species inhabiting hard substrates of the photic zone in the Baltic Sea. Under natural conditions, the depth penetration of *Fucus vesiculosus* increases from the Kattegat towards the central and inner parts of the Baltic Sea (Torn et al. 2006). From being an intertidal species with a narrow distribution belt in the North Sea region, *F. vesiculosus* becomes sublittoral and widens its distribution zone in the Baltic Sea (Waern 1952; von Wachenfeldt 1975). This has been interpreted as a consequence of both physico-chemical and biological changes in growth conditions for the species. The upper limit of the distribution zone thus tends to move downwards as a consequence of the decreasing tidal amplitude and the increasing risk of ice scour during hard winters, while the lower limit of the belt moves downwards as a consequence of decreased competition between macroalgal species at lower salinities (Pedersen & Snoeijs 2001).

The depth limit of bladderwrack seems to be one of the most promising indicators in the Baltic Sea, although there are several difficulties in interpreting the actual monitoring data. The greatest problems exist in the interpretation of the depth limit, since this can be estimated either as the maximum depth distribution of single specimens, the depth distribution of the *Fucus* community (belt) or the depth of optimum growth (maximum coverage).

Data on the depth distribution of *F. vesiculosus* was compiled from published articles and reports as well as from national monitoring databases. Most of the data were from the recent period representing the years 1990–2006, but for a few areas data could be obtained from the 1930s to the 1990s, allowing an assessment of long-term changes.

At present, healthy *F. vesiculosus* vegetation is observed along the Danish, German, Swedish, Finnish, Estonian, Latvian and some Russian coasts. However, this species is almost totally absent from the exposed shores of the eastern Baltic Proper (e.g. the Polish and Lithuanian coasts). The present depth distribution range of *F. vesiculosus* extends from close to the surface down to maximum depths of individual specimens at 1.5–4.5 m on average, depending on the area. The shallowest depth limit of *F. vesiculosus* individuals (about 1.5 m) is recorded in the Kattegat, the Danish Belts and the Sound at the entrance to the Baltic Sea. In the Sound, the average depth limit of *Fucus* individuals is about 2.5 m, while depth limits in the central and inner parts of the Baltic are as deep as 4–5 m on average. At present, the deepest-growing *Fucus* individuals are recorded in the Bothnian Sea and northwestern Baltic Proper at, on average, 5.2 m depth and in extreme cases at 12.2 m depth (Torn et al. 2006; the Swedish monitoring database, unpublished data).

Extensive belts of *F. vesiculosus* are often formed on the hard-substrate bottoms along the coastlines of Estonia, Finland and Sweden. The depth limit of *F. vesiculosus* belts follows the same trend as the depth limit of *F. vesiculosus* individuals, i.e. it increases towards the central and inner areas of the Baltic. The shallowest depth limits for a belt (1.5 m on average) are recorded in Hanö Bay. From here, depth limits increase gradually down to 3 m on average in the Gulf of Finland. When moving to the Bothnian Sea, respective depth limits increase to almost 5.5 m on average. The deepest observations of *F. vesiculosus* belts have been recorded at 7.3 m depth in the Bothnian Sea.

The average depth of maximum coverage of *F. vesiculosus* is at 1–2 m in most areas. Only the Bornholm Basin, the Gulf of Riga and the Bothnian Sea have slightly deeper values (average 2.5 m). The depth of maximum coverage is thus more similar among areas than other indicators of the depth limit.

The best way to translate measurements of indicators for the assessment of eutrophication status is to use the EQR (Ecological Quality Ratio) values calculated from the known reference values for the parameter and actual measurements. Reference values for depth distribution of *F. vesiculosus* are available for sea areas north of the Bornholm Basin. These reference values are derived from national water quality classification systems used for EU Water Framework Directive assessment schemes.

Aggregation of EQR values for the coastal areas evaluated for the assessment period (2001–2006) provides the recent status for this indicator. Only the eastern Baltic Proper and the Gulf of Riga have EQR values over the good/moderate class boundary (acceptable deviation corresponding to the good/moderate boundary has been set at 25% from reference conditions). Other areas show present conditions to be at the moderate level (**Fig. 2.22**). **Fig. 2.23** shows the impact of eutrophication on *Fucus* communities.

Eelgrass *(Zostera marina)*

Eelgrass is a vascular plant of marine origin and its existence in the Baltic Sea challenges its lowest salinity tolerance limit especially in the northern and eastern parts of the Baltic. Although distributed all over the Baltic Sea in areas with salinities above 3, this species reaches its maximum abundance and biomass in the southern part of the sea.

Eelgrass grows markedly deeper along open coasts than in enclosed bays and fjords. This was noted already 100 years ago by Ostenfeld (1908). Moreover, already in the late 19th century, Reinke (1889) noted that eelgrass grows markedly deeper on sandy sediments than on soft bottoms, possibly due to more favourable oxygen conditions on sandy substrate. Moreover, the sandy sediments have more light and are also less easily resuspended than the soft sediments, the both reasons providing a better light climate for eelgrass.

Although differences in salinity do not seem to affect the depth limit of eelgrass, it is most likely that at the species' lower salinity limit in the inner Baltic - where eelgrass mainly grows vegetatively - eelgrass communities re-establish more slowly to the deeper depths (Reusch et al. 1999) than at the entrance to the Baltic, where seeds play an important role in the dispersal of eelgrass (Olesen 1999; Nielsen & Olesen 1994). Another secondary effect of low salinity, which limits the spread of eelgrass, is increasing interspecific competition with freshwater angiosperms, which dominate at the northern range of eelgrass distribution (Baden & Boström 2001). Re-establishment of lost eelgrass beds is therefore likely to take a long time in the inner Baltic Sea.

Eelgrass is a suitable eutrophication indicator especially in the sheltered areas of the southern Baltic (Krause-Jensen et al. 2005). Here, the variation in nutrient loading is able to explain up to 75% of the variation in the eelgrass depth distribution (Nielsen at al. 2002a). The present status of this indicator varies over a large interval depending to a great extent on the specific local conditions. Thus, EQR values for depth distribution of eelgrass can be as low as 0.3 for some areas of the Limfjord (Denmark) and up to 0.89 in some areas of the southern Kattegat (e.g. Isefjord) (**Fig. 2.22**).

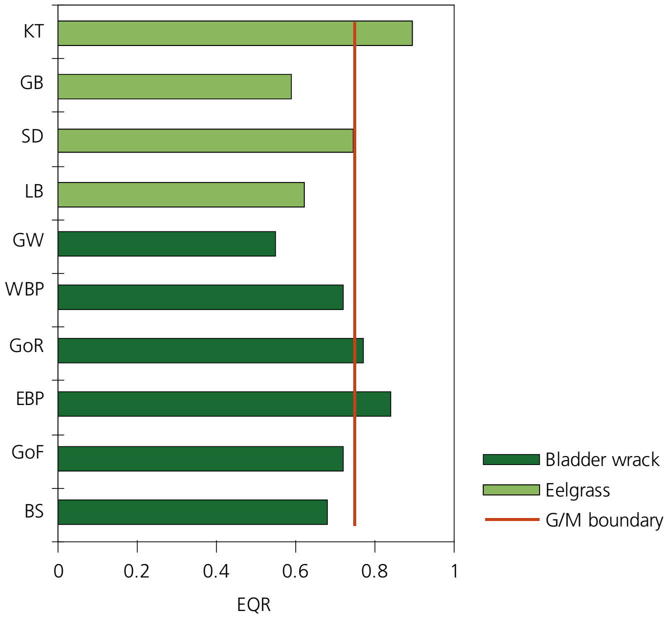


Figure 2.22 EQR values derived from data on maximum depth penetration of eelgrass and bladderwrack. (Reference values obtained from national assessment schemes or values used by the WFD Geographical Intercalibration Group (Baltic GIG)).

Abbreviations: KT = Kattegat, GB = Great Belt, SD = Sound, LB = Little Belt, GW = Western Gotland Basin, WBP = Western Baltic Proper, GoR = Gulf of Riga, EBP = Eastern Baltic Proper, GoF = Gulf of Finland, BS = Bothnian Sea; G/M = good/moderate.

Ratio of annual to perennial species

The ratio of annual to perennial macroalgae and of filamentous algae to *Zostera marina* is a potential indicator for eutrophication because high nutrient concentrations generally favour the growth of ephemeral flora (Sand-Jensen & Borum 1991; Pedersen & Snoeijs 2001).

The group of annual algae includes species that are often opportunistic, as they are favoured by increased nutrient concentrations. They typically have thin tissues capable of rapid rates of nutrient uptake and growth, are relatively short-lived, and represent the so-called r-strategy of population growth. Typical annual and opportunistic algae are, for example, filamentous species such as *Cladophora glomerata*, *Pylaiella littoralis* and *Ectocarpus* sp. and sheet-formed species such as *Ulva lactuca*. In contrast to the annual species, perennial species are slow-growing, often have thicker and more complex tissues, and are represented by, for example, the genera *Fucus*, *Furcellaria* and

Laminaria. Although the classification to annual and perennial is in many cases straightforward, there are common species that are more difficult to define and, therefore, the classification needs to be carefully considered in every case.

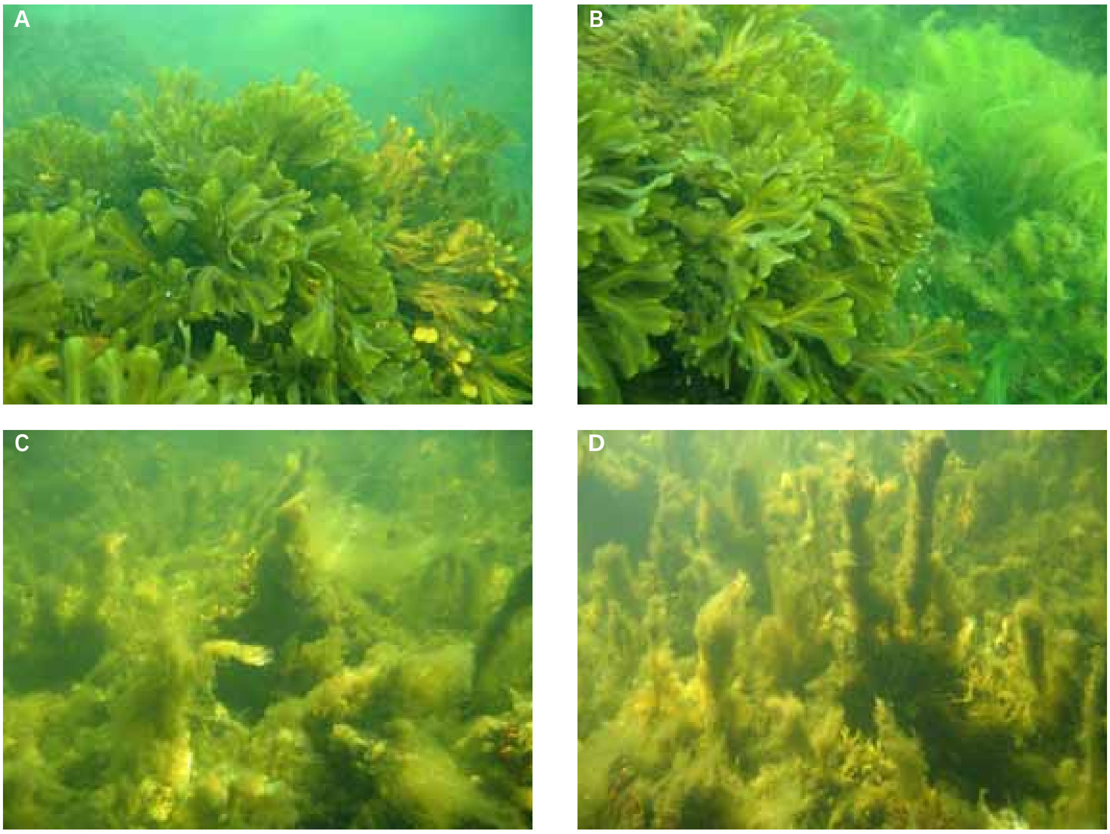


Figure 2.23 ***Fucus*** **sp. along a eutrophication gradient ranging from good conditions (A) to undesirable and severely impaired conditions (D).**

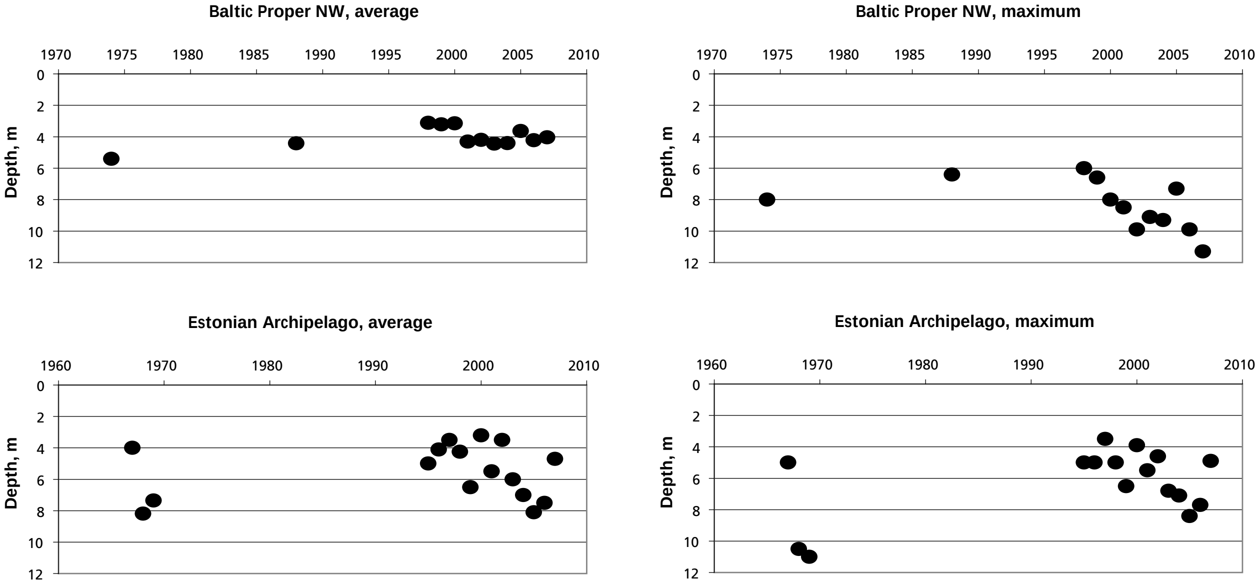


Figure 2.24 **Long-term changes in the distribution of depth limit of** ***Fucus vesiculosus*** **presented as average values for a given year and area and absolute maximum values.**

The relative abundance of annual algae tends to increase as the nutrient loading increases (e.g. Duarte 1995; Korpinen et al. 2007), but only a small amount of information is available on exact relationships between the trophic state of the coastal waters and the amount of opportunistic species. This may in part be due to the fact that the abundance of annual algae depends not only on nutrient concentrations but also on other ecological factors such as grazing, season, fluctuation of sea level, and duration and thickness of ice cover.

Several reports show an increase in the abundance of annual species especially along the eastern coast of the Baltic Proper (Korpinen et. al. 2007). The occurrence of drifting filamentous algal mats is considered to be a problem in the Archipelago Sea, Gulf of Finland and parts of the Gulf of Riga.

Unfortunately, reference conditions are not available for this indicator for most of the coastal areas of the Baltic Sea, although this indicator has a great potential to be used in eutrophication assessment especially in areas where wave exposure or substrate quality sets a limit to the development of species with a larger thallus.

2.5.2 Temporal trends

SAV communities express short- and long-term variability caused by different factors. Short-term, inter-annual variability is usually caused by either seasonal or climatic variations, while long-term changes may often indicate changes in a complex of environmental conditions including eutrophication effects.

Temporal variation of bladderwrack

Information on long-term changes in the depth distribution of *Fucus vesiculosus* in six areas of the Baltic Sea is presented here using the data collected in the EU CHARM project and published by Torn et al. (2006) together with the most recent data obtained from national databases. For all of these areas, data were available since the 1960s and for three areas older data (1930s–1940s) were also found. All areas showed marked declines in *F. vesiculosus* depth distribution over the described period. In the Åland Sea, the decline occurred from the 1940s to the 1950s, while in the remaining areas it occurred between the 1960s/1970s and 1990s. In the eastern Kattegat, the depth limit was 12 m in the 1960s but less than half of that (5 m) in the 1990s. In the Estonian Archipelago, the depth limit of 8 m in the 1960s was reduced to 2–5 m on average in the period 1995–2001. In the northwest Baltic Proper, the depth limit of 5.4 m in 1974 was gradually reduced to around 3 m in the late 1990s. In the Gulf of Finland and the Åland Sea, *F. vesiculosus* penetrated to around 10 m depth in the 1930s/1940s. This depth limit was maintained in the Gulf of Finland until 1975, but was then halved between 1975 and 1994. In the Åland Sea, the decline to around 7 m was found in 1958 and remained at this level until the 1990s. In the Finnish Archipelago Sea, data are available from the 1960s and 1970s and show a decline from 3 m to <2 m during this period. Two areas showed a positive recovery of *F. vesiculosus* depth limit. Thus, in the northwest Baltic Proper, the average maximum depth limit has shifted over 4 m in the period 2001–2007 (**Fig. 2.24**). At the same time, the absolute maximum depths found for *F. vesiculosus* have increased to 8 m in the Estonian Archipelago region in recent years, but not yet reaching the historical records of the mid-1960s (**Fig. 2.24**). Data from the Estonian Archipelago also allowed an assessment of long-term changes in the depth of maximum coverage of *F. vesiculosus*. In 1962–1969, *F. vesiculosus* showed maximum coverage between 1–6 m depth and relatively high coverage down to 10 m depth (Trei 1973). However, data from 1995–2006 show a high coverage (50% and more) only down to less than 3 m.

Temporal trends of eelgrass

Historic data on eelgrass depth limits are available from Denmark, Germany, Poland, and Lithuania. This compilation is based on two publications (Boström et al. 2003; Krause-Jensen et al. 2003).

In Denmark, records of eelgrass depth distribution date back to around 1900 (Ostenfeld 1908). At that time, Danish eelgrass meadows were widely distributed and covered about 1/7 of Danish marine waters. The meadows (except in the low-saline southwestern Baltic waters) were decimated during the wasting disease outbreak in the 1930s but recovered during the following decades. Today, eelgrass occurs again along most Danish coasts but has not reached its former distribution range. It is estimated that the present depth distribution constitutes only about 20–25% of that in 1900. A comparison of historic and recent data sets on depth limits of eelgrass meadows from the same sites showed that around 1900, the depth limit of eelgrass meadows averaged 5–6 m in estuaries and 7–8 m in open waters, while in the 1990s the depth limit of the meadows was reduced by about 50% to 2–3 m in estuaries and 4–5 m in open waters. This large reduction in depth limit can be mainly attributed to impoverished light conditions owing to eutrophication. Depth limits are positively correlated to Secchi depths, which were generally greater around 1900.

In Kiel Bight, the depth limit of eelgrass decreased from 6 m in the 1960s to <2m in the late 1980s probably as a consequence of increasing amounts of shadowing epiphytic filamentous algae (Schramm 1996). In the Greifswald Lagoon, by contrast, the eelgrass populations have remained relatively stable during the period 1930s–1980s (Messner & von Oertzen 1991). In the Gulf of Gdansk, eelgrass grew down to 10 m depths in the 1950s, but was almost totally displaced by filamentous brown algae and *Zanichellia palustris* during the period 1957–1987 (Kruk-Dowgiallo 1996). Along Lithuanian coasts, eelgrass had virtually disappeared before any scientific evaluation was made; eelgrass most likely occurred along the 90-km-long seaside of the Curonian Split covering thousands of hectares, but in the 1990s filamentous green algae dominated the coastal waters and no eelgrass was found (HELCOM 1998).

2.6 Oxygen

The oxygen concentration is a measure of the amount of oxygen dissolved in seawater. It is generally described in terms of the mass, volume or number of moles of oxygen in a litre of seawater. Oxygen content may also be described in terms of oxygen saturation: the ratio of the observed concentration to the theoretical maximum concentration that water of the same temperature and salinity could contain, expressed as a percentage. Oxygen enters seawater from the atmosphere through physical processes, such as wave breaking and diffusion, and is also produced as a byproduct of photosynthesis. As on land, it is necessary for the sustenance of life, except for some types of bacteria that are able to metabolize, for example, sulphur instead. In the Baltic, oxygen is also transported into the system with oxygen-rich inflows from the North Sea, which also reach the deep basins. Because of the strong, permanent stratification in many parts of the Baltic, this abiotic mechanism is most important for oxygenating the deep basins. Large inflows are induced by meteorological events such as a rapid succession of atmospheric depressions. Inflows enter through the Danish Straits and travel rapidly, oxygenating the deep basins. However, there has been a severe reduction in the frequency of large inflows in the past thirty years. The estuarine circulation also causes an inflow of bottom water (baroclinic inflow) which can introduce substantial volumes of oxygen.

Oxygen is consumed by biological activity, based on the utilization of organic matter. Under eutrophication, decomposition of large algal blooms requires large amounts of oxygen, causing the oxygen concentration to fall. Low oxygen concentrations inhibit larval development, and result in the death of fish and benthic communities and the release of nutrients from bottom sediments. In shallow water, these conditions occur most often in the autumn, when the summer phytoplankton bloom decays and the accumulated biomass is degraded (**Fig. 2.25**). Oxygen concentrations also vary diurnally. At night, aquatic plants change from photosynthesis to respiration. In shallow bays, particularly where water exchange, and thus the exchange of dissolved compounds (e.g., dissolved oxygen), is inhibited by a thermocline or by the shape of the bay, respiration may cause hypoxia. Where plant biomass has increased through eutrophication, this hypoxia is exacerbated. This hypoxia is thought to favour certain fish such as stickleback over perch and pike. Stratification hinders the exchange of oxygen between bottom water and the mixed layer, particularly during calm weather periods. If the volume of water below the pycnocline is small, all oxygen in that volume can be quickly consumed. If the oxygen cannot be replaced, either by mixing across the pycnocline or by horizontal advection, hypoxia will develop. This mechanism was the cause of the extensive fish kills observed in the Danish Straits and southern Kattegat in the late summer and autumn of 2002.

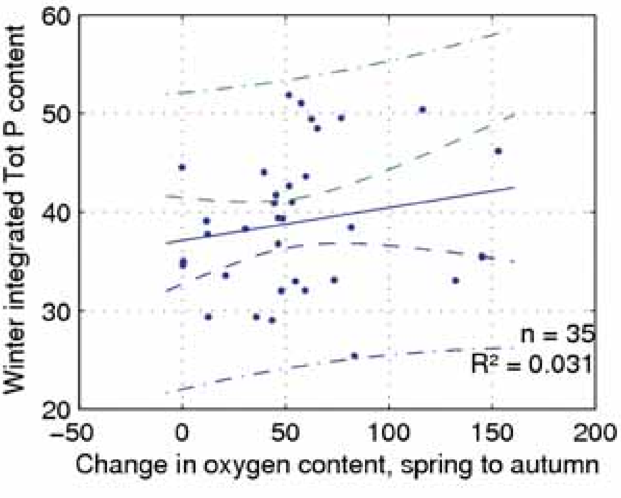
BOX 5: Correlations between oxygen concentrations and nutrient concentrations

Oxygen conditions are both affected by eutrophication and are a driver of eutrophication. When increased nutrient levels lead to increased biological production, oxygen is produced in the surface water - as far as light can penetrate. As this production decays and sinks, however, oxygen is consumed. Oxygen consumption occurs throughout the water column, but it is particularly high around the pycnocline and near the bottom. The relationship between nutrient consumption and the amount of oxygen consumed later is not well defined, however.

As oxygen is consumed, oxygen concentrations fall. Fish cope with these small decreases in concentration by moving up towards the surface or moving away to better-oxygenated regions. Benthic animals have evolved to cope with small changes for short periods. If the decrease in oxygen concentration is great, either because of excessive biological production or because new oxygen is not introduced to the area by physical processes, then the impact of stress on animals becomes apparent, with behavioural changes such as an inability to hunt and eventually death. If the sea bottom is dead for a period, then burrowing organisms no longer oxygenate the sediment, which allows sulphide to move up towards the sediment-water interface. Sulphide releases have been observed in Danish fjords, causing rapid oxygen consumption and fish kills.

If the oxygen concentration continues to fall to close to zero, then the chemical processes affecting nutrient recycling are changed. Phosphate is released in large quantities, becoming available as a further driver of eutrophication. High concentrations of phosphorus relative to nitrogen in brackish waters favour cyanobacteria, which may be toxic. In the Baltic Proper, where there are large anoxic regions, phosphate release from the sediment has given rise to very high phosphate levels at the surface.

It takes time to recolonize areas affected by hypoxia. Small areas can be recolonized fairly quickly, though this is typically by soft-bodied, rapidly reproducing organisms that live on the sediment surface. Long periods with good oxygen conditions are required for communities of mussels and crabs to develop. It is these hard-bodied communities which provide food for demersal fish, such as cod.

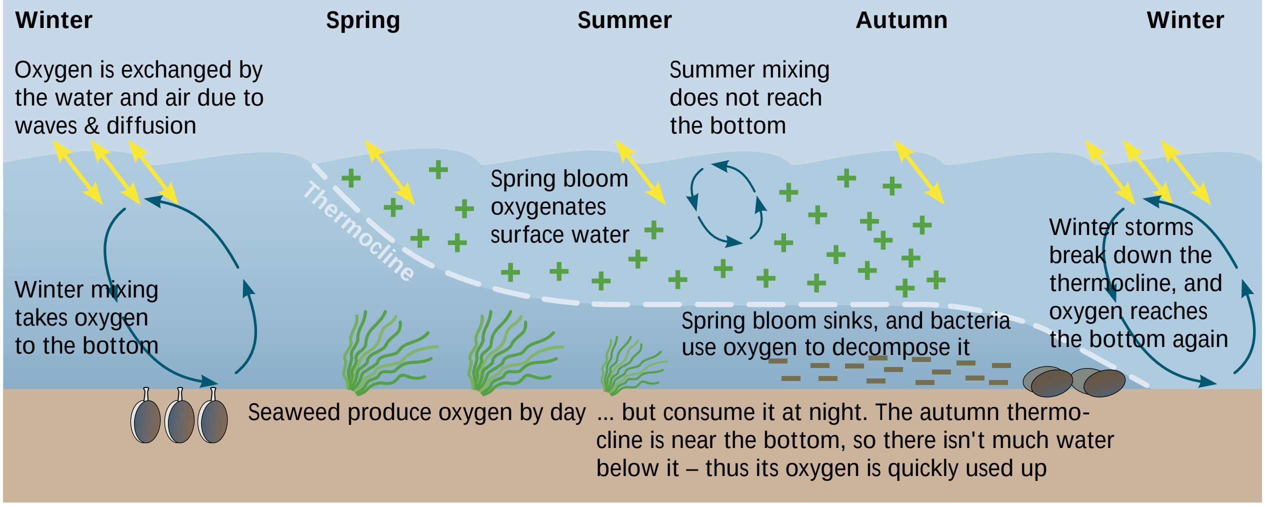


Panel A: The relationship between nutrients and the amount of oxygen consumed is not well defined.

Oxygen conditions in the Baltic are at their best during early spring. Winter storms and cooling lead to well-oxygenated surface water being mixed as deeply as possible. At the onset of spring, primary production leads to super-saturation of oxygen in the surface water. Later in the year, the biomass produced will be decomposed, which consumes oxygen. The amount of oxygen consumed is a function of primary eutrophication processes: large amounts of biomass produced consume large amounts of oxygen. The poorest oxygen conditions occur in the autumn.

Where oxygen concentrations are very low or even zero, bacteria continue to break down organic matter. Under these conditions, they consume what oxygen they can, even chemically bound oxygen within, for example, nitrate. As a by-product of this process, hydrogen sulphide is produced. If oxygen is introduced into this system, it is first consumed in the oxidation of the hydrogen sulphide. Only after the hydrogen sulphide has been oxidized does the oxygen concentration increase. As a result, hydrogen sulphide concentrations are frequently described as ‘negative oxygen’, and it is common practice to plot oxygen concentrations below zero. In the presence of hydrogen sulphide,

Shallow water



Deep water

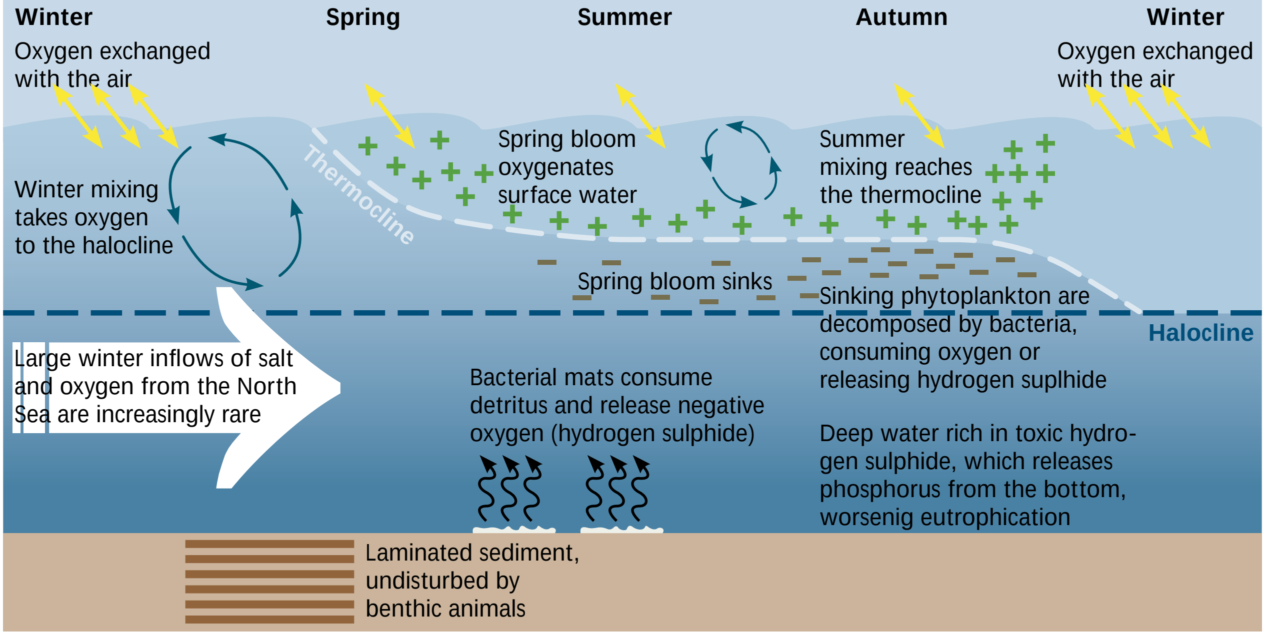


Figure 2.25 Conceptual model of processes affecting oxygen concentrations in shallow and deep water.

the chemical environment becomes reducing, and phosphorus (as phosphate) and silicate are no longer bound to the bottom sediment. With long periods of anoxia, high concentrations of phosphate, silicate and ammonium build up in the (deep) anoxic water. At the end of 2006, concentrations of phosphate in the anoxic deep water of the Eastern Gotland Basin were six times higher than in the surface water. The transport of these nutrients to the surface water would severely exacerbate eutrophication.

The Water Framework Directive considers oxygen to be one of the physico-chemical quality elements that should 'not reach levels outside the ranges established so as to ensure the functioning of the ecosystem'. The HELCOM Baltic Sea Action Plan lists 'Natural oxygen levels' as a distinct objective on the path to achieving the ultimate goal of a 'Baltic Sea unaffected by eutrophication'.

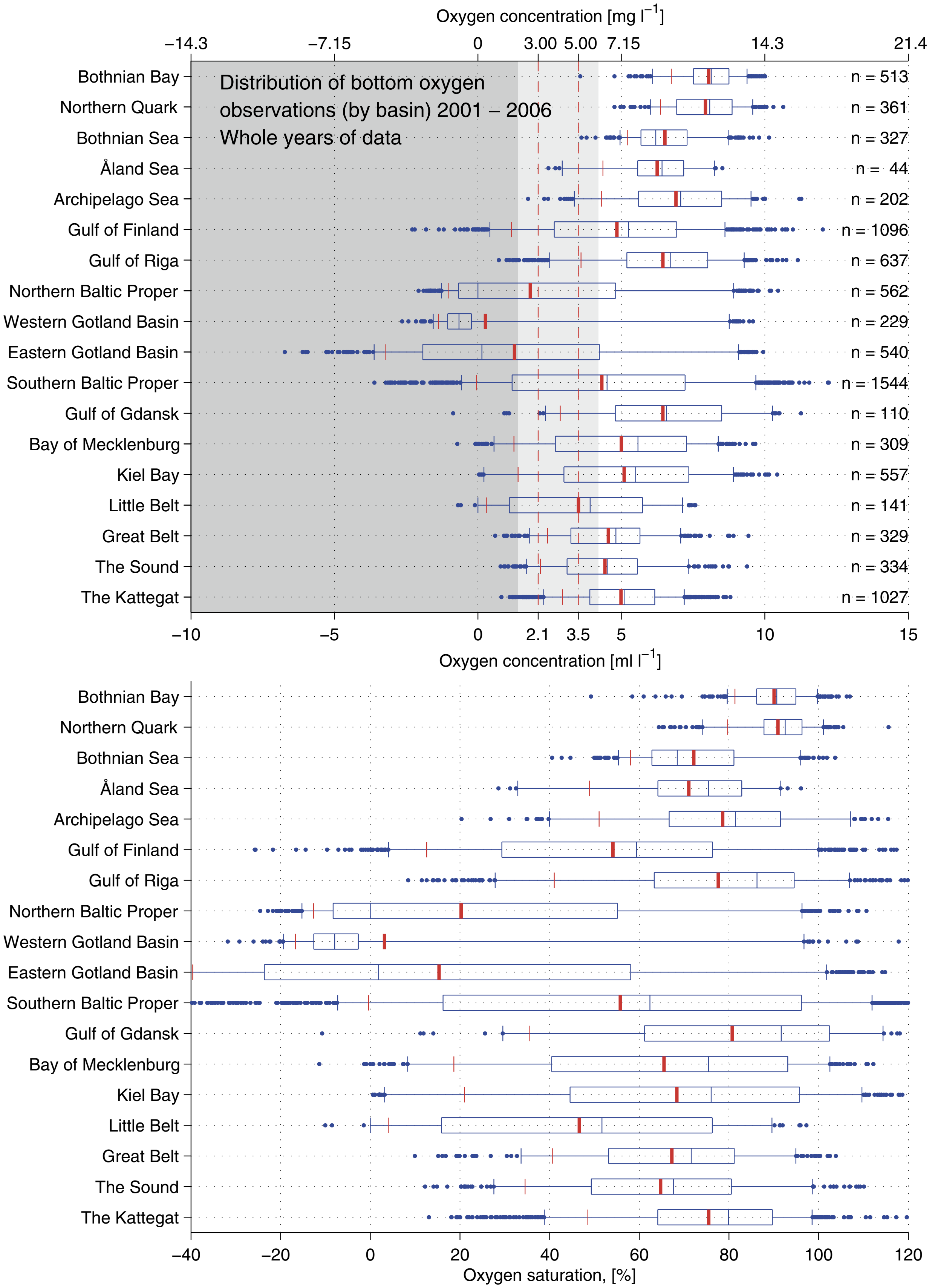


Figure 2.26 Distribution of bottom oxygen data, as concentration (top diagram) and as saturation (bottom diagram). In the top diagram, the dark grey shading identifies near-anoxic and anoxic conditions and the light grey shading shows lower levels of oxygen.

2.6.1 Status 2001–2006

Methods for assessing oxygen status typically rely on oxygen concentrations or saturations known to affect fish, benthic organisms and/or larval development. Studies into these effects have been collated and reviewed by, for example, Diaz & Rosenberg (1995), the U.S. Environmental Protection Agency (US EPA 2000) and Gray et al. (2002). This guidance has been incorporated into assessment criteria, such as the OSPAR Common Procedure (OSPAR 2005, OSPAR 2006), and also legislation, such as the Shellfish Waters Directive, the Bathing Waters Directive and the Urban Waste Water Treatment Directive. Because oxygen concentrations in the Baltic are governed both by natural processes, such as the size and frequency of inflows, as well as by eutrophication effects, a eutrophication assessment methodology consisting of comparing measured oxygen concentrations to a reference level is unsuitable. A suitable method should differentiate between hypoxic events caused by eutrophication and those caused by stagnation.

The methodology adopted uses oxygen (or hydrogen sulphide) concentrations in water samples collected about 1 m above the bottom at a multitude of measurement stations throughout the Baltic. The status metric used was the mean value of the lowest 25% of data observed between 2001 and 2006. Where this metric was above an oxygen concentration of 3.5 ml l^{-1} (equivalent to 5.00 mg l^{-1}, or a saturation of 46.5% at 10°C and salinity of 7.5), the region could have reasonable status. Where the metric did not exceed the 3.5 ml l^{-1} level, it is necessary to determine whether hypoxia is a long-term phenomenon caused by poor water exchange, or whether it is an indication of eutrophication (more commonly associated with seasonal and short-term hypoxia).

In the Gulf of Bothnia, including the Archipelago Sea and the Åland Sea, oxygen status appears to be good (**Fig. 2.26**). In the Bothnian Bay, and the Bothnian, Archipelago and Åland Seas, a few outlying observations fall below the 3.5 ml l^{-1} threshold. The status in the Gulf of Riga is just above the threshold. The status in the Gulf of Finland, Northern Baltic Proper, Western and Eastern Gotland Basins and southern Baltic Proper (including the Bornholm and Arkona Basins) falls well below the threshold, as does the status in the Bay of Mecklenburg, Kiel Bay and the Little Belt. The Gulf of Gdansk, Great Belt, Sound and Kattegat show an oxygen status below the 3.5 ml l^{-1} threshold, but exceeding 2.1 ml l^{-1}.

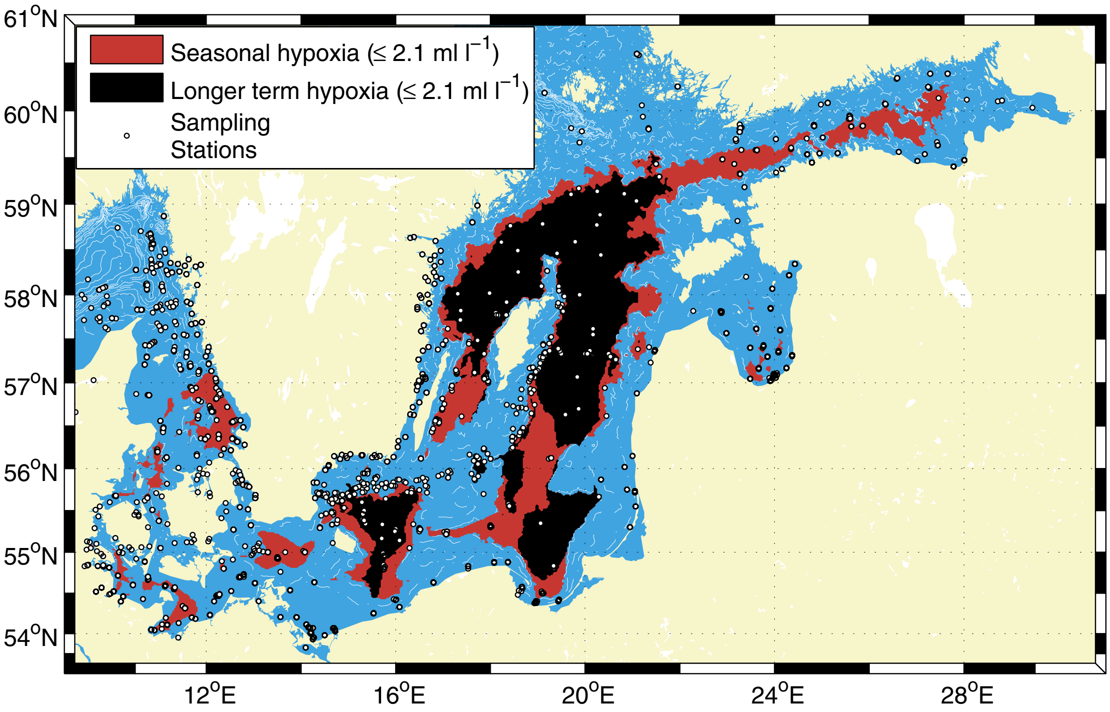


Figure 2.27 **Extent of seasonal hypoxia (red) and longer-term hypoxia (black) during 2001–2006. Long-term hypoxia occurs throughout the year.**

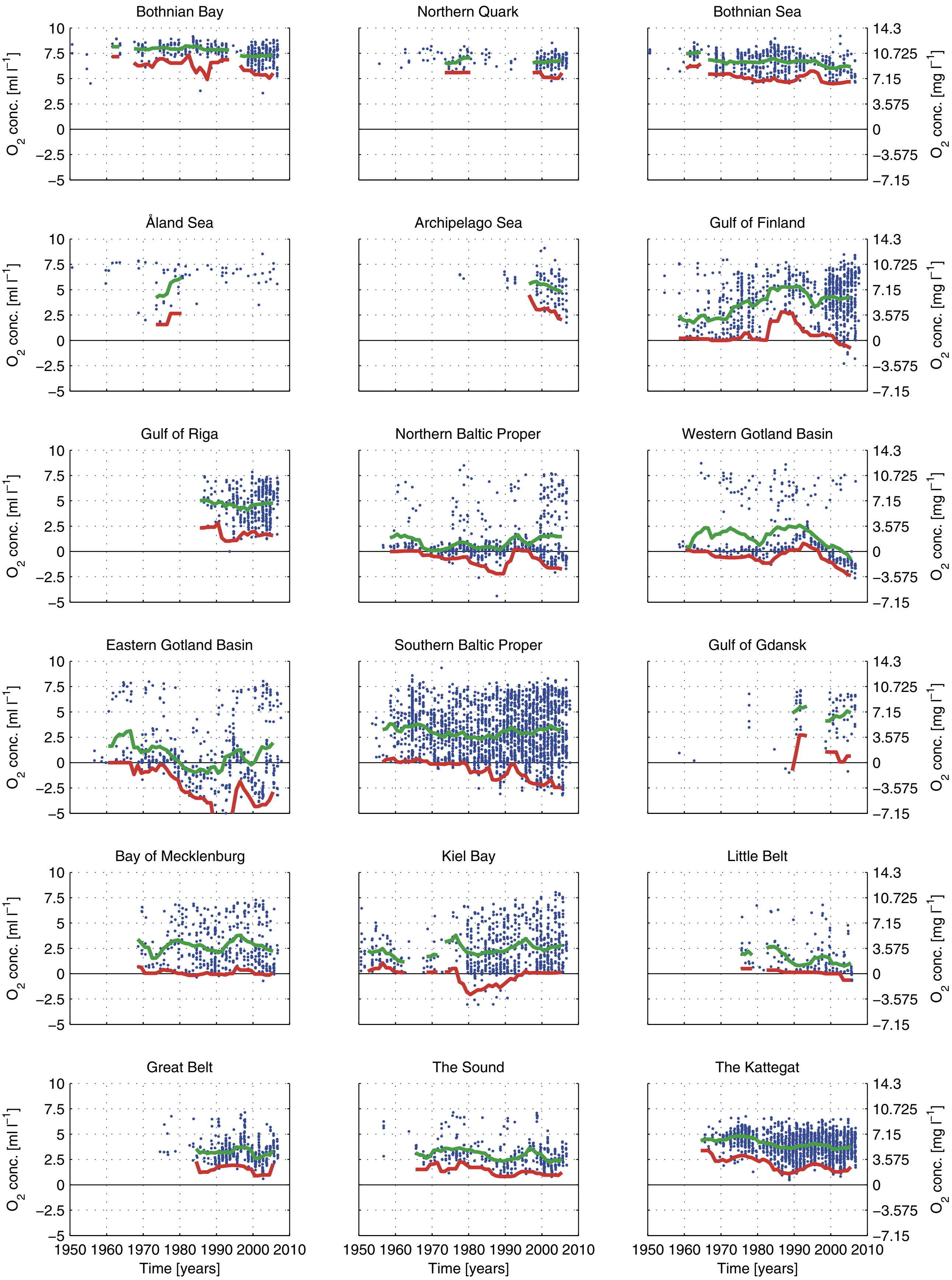


Figure 2.28 Time series of autumn bottom oxygen concentrations, 1950–2006. The green line is the 5-year running mean; the red line is the 5-year running mean of the lowest 25% of the data.

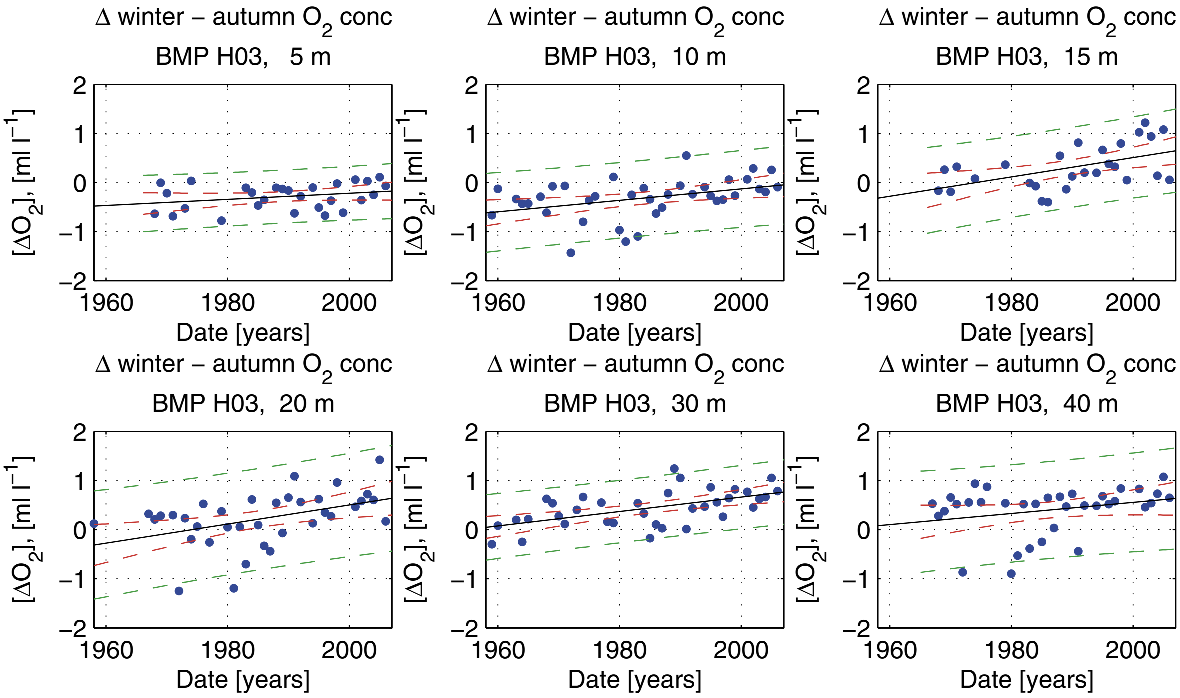


Figure 2.29 **Increases in the difference between winter and autumn oxygen concentrations at a station in the Northern Baltic Proper at different depths. Increases in the difference between winter and autumn oxygen concentrations at station BMP H03 in the Northern Baltic Proper at different depths ranging from 5 to 40 m between 1958 and 2007.**

The extent of seasonal hypoxia was determined by studying data from late summer and autumn (August, September and October) between 2001 and 2006 and comparing them with data collected during winter and spring (January to May). During late summer, hypoxia is at its maximum extent. In winter, only the regions affected by stagnation remain hypoxic. The difference between these areas indicates regions affected by seasonal hypoxia, a strong indication of eutrophication pressure. The extent of the regions affected by seasonal and long-term hypoxia is shown in **Fig. 2.27**.

Seasonal hypoxia (bottom concentrations below 2.1 ml l^{-1}) occurred in the southern Kattegat, the Sound, Little Belt, Kiel Bay, Bay of Mecklenburg and Arkona Basins. Seasonal hypoxia in the Great Belt has occurred, but is very patchy. From the Bornholm Basin through the Baltic Proper to the Gulf of Finland, seasonal hypoxia occurs below about 70 m depth. Long-term hypoxia occurs in the deepest part of the Bornholm Basin, the Gdansk Deep, the Eastern and Western Gotland Basins and in the Northern Baltic Proper. In the Gulf of Finland and Gulf of Riga, hypoxia is almost all seasonal. In mild years, autumn hypoxia may extend through the winter. No hypoxia, either seasonal or long term, was observed in the Gulf of Bothnia (so these areas are not shown in **Fig. 2.27**).

2.6.2 Temporal trends

Time series of autumn bottom oxygen conditions from each basin have been plotted (**Fig. 2.28**). A five-year running mean was fitted to the data, as was the five-year running mean of the lowest 25% of the data, to indicate changes in the occurrence of extremes. Data have been interpreted where more than ten observations have been available in the 3-month period.

To indicate the annual consumption of oxygen, the difference between the winter and the following summer's oxygen concentration was calculated, after normalizing the data to a common temperature and salinity. Under the scenario of increasing eutrophication, it would be expected that there would be an increasing difference between winter and summer concentrations. In the Northern Baltic Proper, this was found to be the case in the upper 40 m (**Fig. 2.29**). In the Kattegat and the Sound, the extra consumption appeared to be limited to the upper 15 m.

Gulf of Bothnia

In the Bothnian Bay, the mean of the lowest 25% bottom oxygen concentrations has decreased from more than 7 ml l^{-1} until the end of the 1990s to below 6 ml l^{-1} in 2006. Only part of this change is reflected in the oxygen saturation data, suggesting that it is partly due to changes in hydrography, particularly temperature. In the Bothnian Sea, there was a decrease in both concentrations and saturation from 1970 until 1995, after which both concentration and saturation have remained stable. Sufficient data are only available from the Archipelago Sea since 1998. These data show a rapid decrease in concentration and saturation. The most recent values are below the 3.5 ml l^{-1} threshold.

Gulf of Finland

The lowest 25% of the data remained almost constant from 1960 to 1980, at close to 0 ml l^{-1}. In the mid-1980s, there was a rapid increase in concentration, which peaked around 1990. Since 1990, the lowest 25% concentrations have decreased from around 3.5 ml l^{-1} to below 0 (hydrogen sulphide) as anoxic water from the Baltic Proper has affected the western areas. The mean bottom oxygen concentration has remained fairly constant, however, possibly due to weaker stratification allowing more mixing in the remainder of the Gulf, and thus preventing stagnation.

Gulf of Riga

Data are available from the end of the 1980s, although they are sufficient only from 1994. During this period, autumn means (lowest 25%) ranged from 1.8 to 4.8 ml l^{-1} without an apparent trend.

Northern Baltic Proper, Eastern and Western Gotland Basins

The records show the development of the long stagnation period, which ended with the inflows of the early 1990s. This is particularly apparent in the Eastern Gotland Basin, where the mean of the lowest 25% decreased from −0.3 ml l^{-1} in 1969 to −5.6 ml l^{-1} in 1992. Oxygen concentrations have worsened again, owing to the present stagnation period. In the Northern Baltic Proper, the worsening conditions were apparent in the mean of the lowest 25% of the data, but not in the overall mean of autumn bottom oxygen.

Southern Baltic Proper and Gulf of Gdansk

The mean bottom oxygen concentrations in the southern Baltic Proper have remained fairly constant, as many stations in the basin are shallow, with good water exchange. The mean of the lowest 25% has shown a decrease of about 2.5 ml l^{-1} since 1960. The 1993/1994 inflows were apparent, but concentrations have declined since then. In the Gulf of Gdansk, mean concentrations appear to have increased recently, although the time series is too short to interpret.

Figure 2.30 Bottom water sampler (panel A) and free sulphide at the surface of the water in Odense Fjord, released because of hypoxia (panel B).

Bay of Mecklenburg and Kiel Bay

Data were available from 1970 in the Bay of Mecklenburg, and from the 1950s in Kiel Bay. The mean of the lowest 25% has been constant in the Bay of Mecklenburg. An increase in concentration and saturation occurred in the second half of the 1990s, but this change was short-lived. In Kiel Bay, concentrations were similar, except during the late 1970s and 1980s, when hydrogen sulphide was present. It is not clear whether the change since 1990 is real or due to a change in sampling (no longer sampling for hydrogen sulphide).

Little Belt, Great Belt and Sound

Oxygen concentrations appear to be worsening in the Little Belt, although this may be an apparent change owing to changes in sampling (**Fig. 2.30**). Conditions are better in the Great Belt and in the Sound. Both regions showed an improvement at the end of the 1990s, but this was not sustained beyond 2000. There was a decrease in both concentration and saturation in the Sound from the end of the 1970s until 2000, after which levels have been stable.

Kattegat

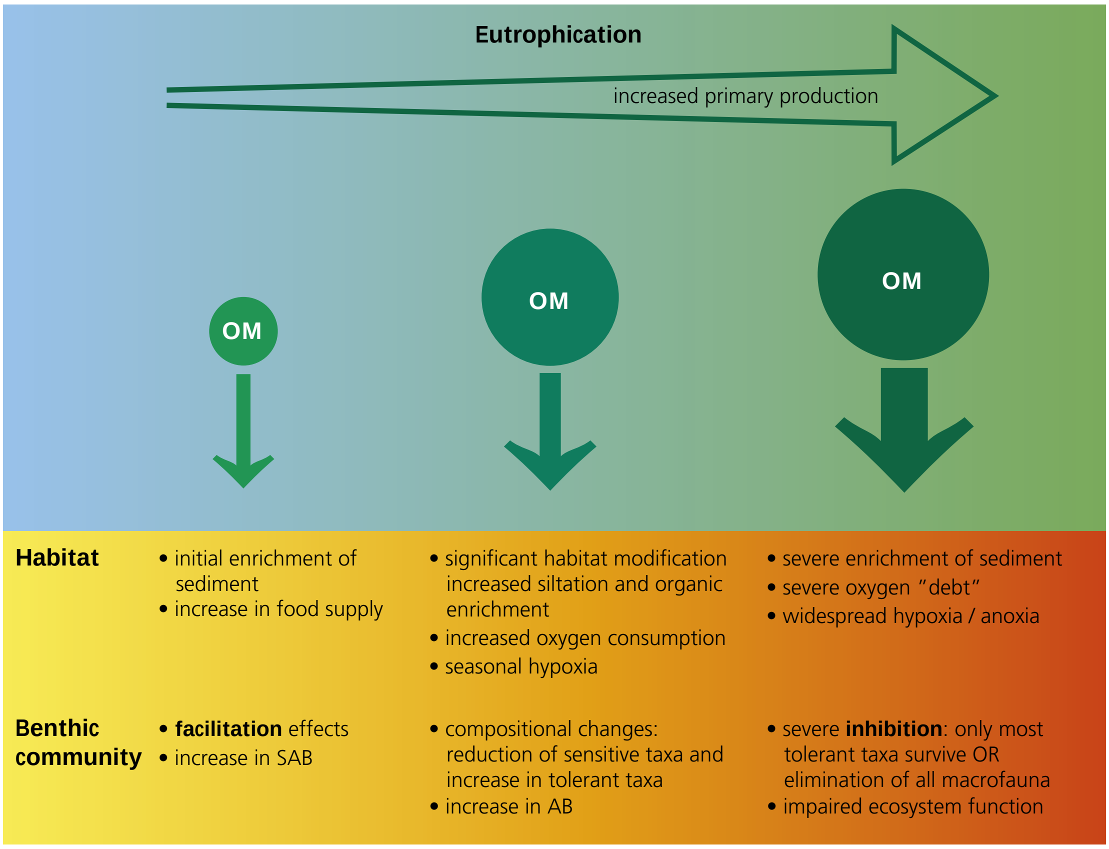
Both mean and mean-of-the-lowest-25% concentrations have decreased in the Kattegat since the start of the 1970s. The decrease in saturation has also been substantial: from 50% in the late 1960s to close to 20% at the end of the 1980s. Since 1990, concentrations and saturations have stabilized.

2.7 Benthic invertebrate communities

Benthic invertebrate communities are good indicators of environmental status. Owing to their relative longevity (years to decades), the composition of benthic communities integrates environmental conditions over longer periods of time. Hence, variations in environmental characteristics, such as salinity, oxygen, food supply, biotic interactions, and different types of disturbances (both natural and anthropogenic), are reflected in the composition of communities in time and space. While physiological tolerances to salinity and oxygen may play the most important role in defining the distribution of species, macrobenthic communities are generally food limited (Pearson & Rosenberg 1978) and the abundance and biomass of benthic invertebrates correlates to some extent with the deposition of pelagic organic material (Josefson & Conley 1997). Benthic environments are at the receiving end of the accumulation and burial of organic material, and healthy benthic communities play an important role in benthic-pelagic coupling and in the mineralization of organic matter settling on the seafloor. Benthic communities generally respond to organic enrichment in a predictable manner. Initial stimulatory effects on benthos are gradually replaced by degradation of communities as eutrophication advances. Increasing organic enrichment and bottom-water hypoxia and anoxia alter benthic community composition by reducing sensitive species and increasing the proliferation of tolerant species. At advanced stages of eutrophication, significant reductions in diversity and ecosystem function accompany these compositional changes.

The relationship between macrobenthic communities and eutrophication in the Baltic Sea needs to be gauged against the strong environmental gradients that provide the framework for species distributions. The latitudinal distribution of macrozoobenthos in the Baltic Sea is limited by a gradient of decreasing salinity. The decreasing salinity reduces macrozoobenthic diversity, affecting both the structure and function of benthic communities (Elmgren 1989; Rumohr et al. 1996; Bonsdorff & Pearson 1999). In addition, the distribution of benthic communities is driven by strong vertical gradients. Generally, the more species-rich and abundant communities in shallow-water habitats differ from the deep-water communities, which are dominated by only a few species (Andersin et al. 1978). The Baltic Proper has a more or less permanent halocline at 60–80 m, whereas in the Gulf of Bothnia stratification is weak or absent. The halocline in deeper waters and seasonal pycnoclines in coastal waters restrict water exchange, which may result in oxygen deficiency and a severe reduction or complete elimination of macrozoobenthic communities.

As an indirect indicator of eutrophication, macrozoobenthos does not respond directly to causative factors such as increased levels of nutrients.



Habitat	• initial enrichment of sediment • increase in food supply	• significant habitat modification increased siltation and organic enrichment • increased oxygen consumption • seasonal hypoxia	• severe enrichment of sediment • severe oxygen "debt" • widespread hypoxia / anoxia
Benthic community	• **facilitation** effects • increase in SAB	• compositional changes: reduction of sensitive taxa and increase in tolerant taxa • increase in AB	• severe **inhibition**: only most tolerant taxa survive OR elimination of all macrofauna • impaired ecosystem function

Figure 2.31 A conceptual model describing the relationship between increasing deposition of organic matter (OM) and changes in soft-sediment habitats and macrobenthic communities.
S = species, A = abundance, B = biomass.

Thus, while macrobenthic community composition provides an excellent measure of environmental status, it is more difficult to ascertain and quantify functional relationships to eutrophication. Pearson & Rosenberg (1978) qualitatively described macrozoobenthic responses to increased organic loading. Several attempts have been made to quantify functional relationships for this successional model, e.g. for oxygen (Gray et al. 2002) and organic carbon (Hyland et al. 2005). Initial positive effects of eutrophication and organic enrichment on food-limited benthic communities are reflected as higher abundances and biomasses. This pattern is well documented in coastal areas such as the Åland archipelago (Bonsdorff et al. 1997a, 1997b) and also in the open Baltic Sea above the halocline (Cederwall & Elmgren 1990). For example, in the Bothnian Bay, where the background concentrations of nutrients are relatively low, positive relationships between increased nutrients in the water column and benthic communities are observed. Increasing amounts of nutrients result in a surplus of organic material reaching benthic habitats. This is not tolerated by sensitive, large-sized and long-lived species and the increase in organic enrichment and subsequent disturbance will initially be seen as large fluctuations in benthic diversity, abundance and biomass. Species composition will change as conditions deteriorate, and the advantage gained by smaller-sized, tolerant species will result in decreasing total biomass and diversity of the benthic community. At advanced stages of organic enrichment, most bottom-water oxygen is consumed by the decomposition of organic material (mainly due to bacteria), resulting in hypoxia and anoxia and initiating the release of toxic hydrogen sulphide from the sediments. At these advanced stages of hypoxia and anoxia, macrozoobenthos is eliminated and important ecosystem services are lost (**Fig. 2.31**).

Perhaps the single strongest factor influencing the biodiversity of benthic communities is the increased prevalence of oxygen-depleted deep water. Hypoxia has resulted in habitat loss and the elimination of benthic macrofauna over vast areas and has severely disrupted benthic food webs. Therefore, separating naturally occurring hypoxia from eutrophication-induced hypoxia is important and needs consideration when assessing eutrophication and benthic invertebrate responses. While hypoxia is to some degree a natural phenomenon in the Baltic, it is also clear that the spatial and temporal extent of oxygen deficiency has increased over the past decades due to eutrophication (Karlson et al. 2002; Diaz & Rosenberg 2008). Although the eutrophication-induced oxygen depletion has not been quantified for the deep bottoms in the Baltic Sea, a link between eutrophication, physical parameters, oxygen deficiency and benthic fauna has been shown, e.g., for Danish coastal and open sea areas, where nitrogen loading from land was demonstrated as a crucial factor for the development of hypoxia (Conley et al. 2007).

The assessment: The benthic invertebrate assessment for coastal waters was produced by a group of national experts from Contracting Parties. It included setting reference conditions and identifying the status for the period 2001–2006. The assessment was typically based on several stations per area or water body. However, some discrepancies were found in the amount and quality of data available for the assessment. For example, coastal data have usually been gathered for smaller water bodies, which have been pre-defined by environmental conditions. This does not always match the sea-area specific approach of this assessment. This limits the quality and interpretation of the assessment. The Gulf of Riga was not assessed as no quantitative data were obtained. Assessed parameters have been defined separately for each country, according to the EU Water Framework Directive (WFD) (Anon. 2000).

Owing to the small number of species, strong gradients in species diversity and the potential for species sensitivities to differ between regions, it is likely that no single index or measure would be applicable or suitable as an assessment tool for all regions of the Baltic Sea. Hence, for coastal waters, each country has conducted its own evaluation of suitable measures to use when assessing the state of the benthic community. Methods and parameters therefore vary. Generally, the criteria of the WFD have been fulfilled, and hence benthic abundance, composition, and the proportion of tolerant and sensitive taxa to disturbance have been incorporated. The WFD sets clear instructions on how to assess benthic communities in coastal areas, while currently no procedures have yet been developed for the open Baltic Sea, because the Marine Strategy Framework Directive became effective only recently and the implementation process has not yet started.

Even though several different indices and parameters are included in this benthic assessment, there has been a general understanding that there are neither existing reference communities nor reference values for the benthic fauna in the Baltic Sea. The Baltic Sea is a young ecosystem still undergoing post-glacial succession and is very dynamic on decadal time scales; hence, decadal time-scale fluctuations in salinity regimes and consequent changes in benthic communities shift the baseline for assessing reference conditions. In addition, historical data are scarce, and the reference value has often been set as ‘the best possible’ value obtained through expert judgement when analysing existing data.

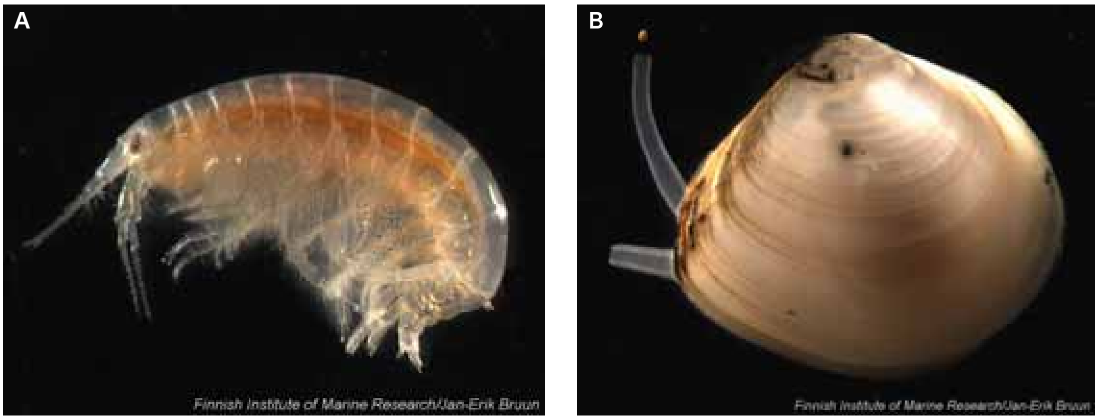


Figure 2.32 ***Monoporeia affinis*** **(panel A) and** ***Macoma balthica*** **(panel B).**

For open-sea areas, the assessment is principally based on a combination of HELCOM data and long-term monitoring data collected by the Finnish Institute of Marine Research since 1965. Additional data from Sweden were obtained for the Bornholm and Arkona Basins. A new indicator was developed to provide a harmonized assessment of benthic invertebrate status in the open-sea areas across all major sub-basins. This indicator is simply based on the average benthic invertebrate diversity in a sub-basin where reference conditions and acceptable deviation have been derived utilizing the best available data sets from 1965–2006. While ideally a benthic invertebrate eutrophication indicator should include community abundance and biomass and those developed for the WFD may be adequate and informative for coastal waters, they are not satisfactory for a broad-scale and harmonized analysis across all open-sea basins. In open-sea areas, species diversity is generally substantially lower than in coastal waters (except for the Kattegat) and natural abundance fluctuations by dominant species are large (e.g. the amphipod *Monoporeia affinis* in the Gulf of Bothnia, **Fig. 2.32**). Hence, separating eutrophication effects from natural fluctuations would be very complex to include in a quantitative eutrophication indicator for benthic invertebrates. Species diversity is less sensitive as a measure but appears to work adequately for assessing benthic invertebrate status in open-sea areas (HELCOM 2009).

2.7.1. Status 2001–2006

In this status assessment, Ecological Quality Ratios (EQRs) have been used to evaluate benthic invertebrate status in open and coastal areas. The EQR values represent an average for the assessment time period of 2001–2006. In principle, the EQR approach should enable comparisons between intrinsically different water bodies. However, area-specific acceptable deviations from reference conditions ultimately define the acceptable status for each area, although they are not discussed here. Generally, it appears that EQR values are comparable; however, in the case of, for example, the BQI index (Benthic Quality Index, Rosenberg et al. 2004; Blomqvist et al. 2006), very large acceptable deviations (up to 86%) have been tolerated and hence even exceedingly low EQR values may be above the good/moderate (G/M) border. Thus, when comparisons of EQR values are made between, e.g. western and eastern coastal waters of the Bothnian Bay, it should be kept in mind that the status (i.e. below or above the G/M border) may be comparable despite large differences in EQR. Nevertheless, these discrepancies highlight the problem of using different indices in different countries and the evident need for careful inter-calibration. Temporal trends in benthic invertebrate status have been assessed for open-sea areas only.

Open sea

Benthic invertebrate diversity and, therefore, reference conditions differ markedly between sub-basins owing to the gradient in salinity, which constrains species distributions (**Fig. 2.33**). A total of eight basins were evaluated and the reference conditions, measured as the average number of species, varied between 18.3 in the Arkona Basin and 2.0 in the Bothnian Bay. For the years 2001–2006, benthic invertebrate status varied considerably between sub-basins and was related to the widespread occurrence of hypoxia and anoxia in the Baltic Proper and the Gulf of Finland (**Fig. 2.33**). None of the sub-basins can be regarded as pristine and even the Gulf of Bothnia, where EQR values were the highest at 0.83, showed a 17% reduction from defined reference conditions. The entire Baltic Proper, from the Bornholm Basin to the northern Baltic Proper and the Gulf of Finland, was in a severely disturbed state (**Fig. 2.33**).

In the open Kattegat, EQR values were significantly above the G/M border of 0.63 and benthic invertebrate status was therefore acceptable (**Table 2.1**). Although this area was also affected by the severe hypoxia in 2002 (Hansen et al. 2003), faunal reductions only occurred in a few local areas and the subsequent recolonization was rapid.

The Arkona Basin is regularly flushed by saltwater inflows; the average EQR value for the assessment period was 0.76 (range: 0.55–1.09) and above the G/M border (**Fig. 2.33**), indicating no or only slight deviations from reference conditions. This was supported by a Danish evaluation, which also indicated acceptable status. However, in the Bornholm Basin conditions were severely disturbed, with an average EQR value of 0.24 (range: 0.12–0.40), representing a 76% reduction from defined reference conditions. Also in the southeastern Gotland basin and the northern and central Eastern Gotland

Basin, conditions were severe, with EQR values of 0.23 (range: 0.12–0.75) and 0.12 (range: 0.05–0.17), respectively (**Fig. 2.33**). No benthic fauna was recorded in the Gdansk Deep during 2001-2006, resulting in an EQR of 0.00.

In the northern Baltic Proper, no fauna whatsoever was recorded for the assessment period, resulting in an EQR value of 0. This reflects the consistently bad oxygen conditions in this open-sea area. Conditions in the Gulf of Finland were more variable, but also there benthic invertebrate status was bad, with an EQR of 0.39 (range: 0.16–0.58). In the eastern parts of the Gulf of Finland (Russian open-sea waters), conditions were below the G/M border, with an average EQR of 0.36 (**Table 2.1**).

In the Gulf of Bothnia, where water column stratification is weak and oxygen conditions are generally good, EQR values were above the good/moderate border, i.e. an average of 0.83 in both the Bothnian Sea (range: 0.69–1.0) as well as in the Bothnian Bay (range: 0.7–0.94) (**Fig. 2.33**).

Coastal waters

A summary of reference conditions, EQRs for the assessment period (including average/median and the range) as well as the G/M border, which is defined by the acceptable deviation, is presented in **Table 2.1**. When assessing the status for the Danish areas, including the open Kattegat, a Wilcoxon signed ranks test was used to evaluate whether data assessed using the multi-metric DKI index (Borja et al. 2007) were above or below the G/M border.

Danish straits and sounds

The status of Danish coastal waters in the Belt Sea, including Aarhus Bay, north of Funen and southern Little Belt, showed an acceptable benthic invertebrate status during 2001–2006 using a G/M border of EQR = 0.53. However, using the G/M border estimated for the deep Kattegat (EQR = 0.63), the status was not acceptable. The Belt Sea area is heterogeneous, with different salinity regimes, and there is no single G/M border that can be applied there. In the

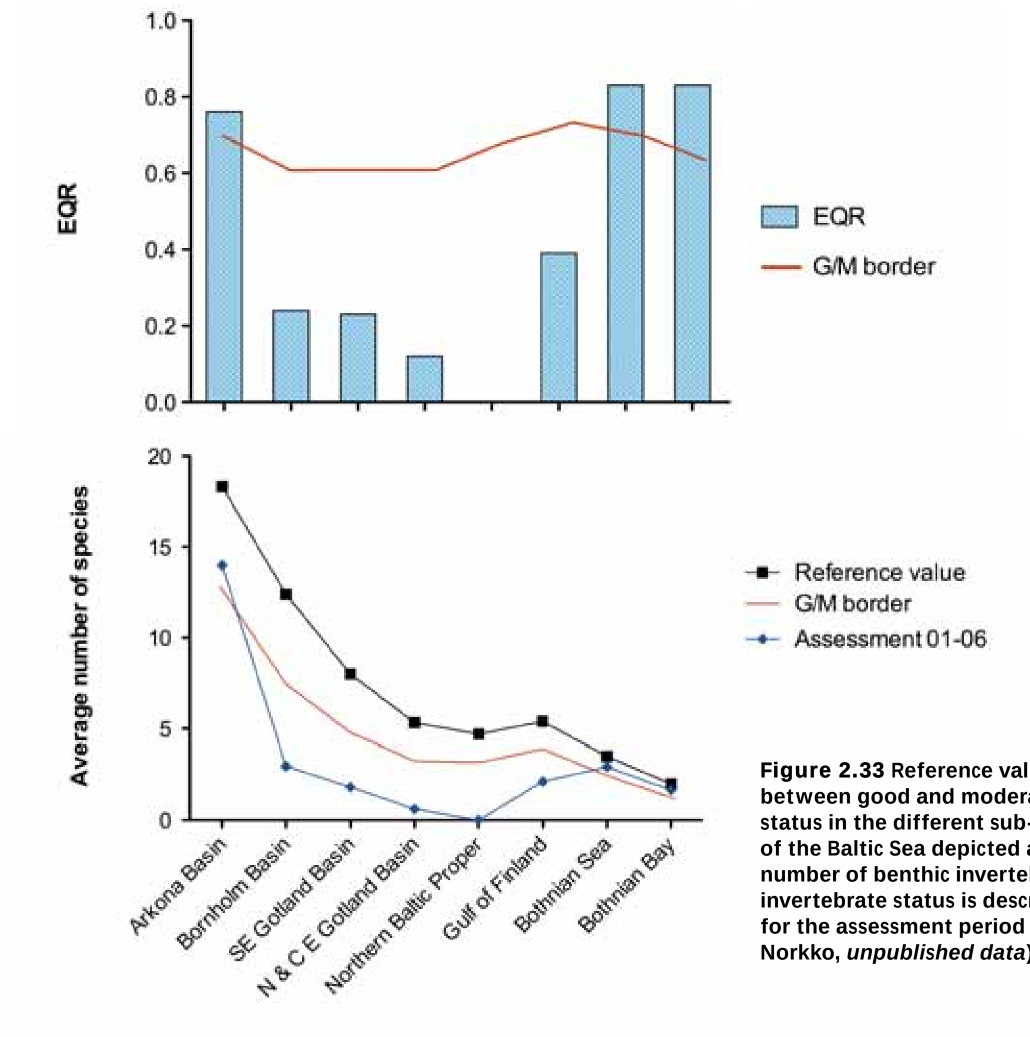


Figure 2.33 **Reference values and the border between good and moderate (G/M) ecological status in the different sub-basins in open-sea areas of the Baltic Sea depicted as EQR and the average number of benthic invertebrate species. Benthic invertebrate status is described as an average for the assessment period 2001–2006 (Villnäs & Norkko, *unpublished data*).**

Table 2.1 **Summary of reference conditions, EQRs for the assessment period (including average/median and the range) as well as the good/moderate (G/M) border for coastal areas of the Baltic Sea. The G/M border is defined by the acceptable deviation (AcDev).**

Area	Parameter	Reference condition	EQR average (*median)	EQR-range	G/M border	Acceptable deviation (%)
Kattegatt, open parts	DKI	1	0.71*	0.44-0.88	0.63	37
Danish Strait, Belt Sea	DKI	1	0.55*	0.23-0.79	0.53-0.63	37-47
Danish Strait, The Sound	DKI	1	0.74*	0.64-0.80	0.63	37
Arkona basin, open parts	DKI	1	0.58*	0.39-0.68	0.37	63
Bornholm, western Hanö Bight	BQI	1	0.33	0.24-0.43	0.25-0.29	71-75
Bornholm Deep	Presence/Absence	Presence	0,00	-	-	100
Kiel Bight, northwestern	MarBIT	1	0.58*	0.47-0.61	0.60	40
Kiel Bight, southwestern	MarBIT	1	0.50*	0.18-0.71	0.60	40
Mecklenburg Bight, Fehmarnbelt	MarBIT	1	0.62*	0.57-0.71	0.60	40
Mecklenburg Bight	MarBIT	1	0.59*	0.55-0.64	0.60	40
Southern Baltic Proper, Pomeranian Bay, central Polish coast	BQI	1	0.47-0.67	-	0.46	54-66
Southern Baltic Proper, internal Gulf of Gdansk	BQI	1	0.67	-	0.46	54
Southern Baltic Proper, Gdansk Deep	Presence/Absence	Presence	0,00	-	-	100
Eastern Baltic Proper, Lithuanian coastal waters	Average no of sp	1	0.69	0.47-0.84	0.70-0.83	17-30
Eastern Baltic Proper, Gulf of Riga & Pärnu Bay	ZKI	1	0.5-0.8	0.50-0.80	0.50	50
Eastern Baltic Proper, Estonian western coastal waters	ZKI	1	-	0.30-0.80	0.50	50
Western Baltic Proper, Kalmarsund	BQI	1	0.31	0.29-0.33	0.29	71
Western Baltic Proper, northern Gotland coastal waters	BQI	1	0.35	0.27-0.43	0.33	67
Western Baltic Proper, western Gotland coastal waters	BQI	1	0.26	0.08-0.49	0.25-0.40	60-75
Western Baltic Proper, mainland coastal waters	BQI	1	0.31	0.14-0.58	0.25-0.30	70-75
Archipelago Sea and the western Gulf of Finland	BBI	1	0.55	0.00-1.02	0.32-0.44	42-47
Gulf of Finland, Estonian coastal waters	ZKI	1	0.50-0.80	0.50-0.80	0.50	50
Gulf of Finland, Russian outer coastal waters	Abundance of *M. affinis*	1	0.36	-	0.5	50
Gulf of Finland, Finnish coastal waters	BBI	1	0.29	0.02-0.62	0.42-0.56	44-58
Åland Sea, Swedish coast	BQI	1	0.20	0.11-0.36	0.29-0.31	69-71
Bothnian Sea, eastern coastal waters	BBI	1	0.65	0.44-0.76	0.56	44
Bothnian Sea, western coastal waters	BQI	1	0.26	0.04-0.47	0.29-0.31	69-71
The Quark, eastern coastal waters	BBI	1	0.62	0.29-0.86	0.57-0.58	42-43
The Quark, western coastal waters	BQI	1	0.18	0.11-0.28	0.14-0.31	69-86
Bothnian Bay, eastern coastal waters	BBI	1	0.69	0.42-1.07	0.56-0.58	42-44
Bothnian Bay, western coastal waters	BQI	1	0.13	0.02-0.31	0.14-0.30	70-86

All abbreviations related to the parameters assessed can be found in the Glossary on page 133.

Sound, the EQR values were above both estimates of the G/M border, suggesting acceptable status.

Kiel Bight

The northwestern Kiel Bight exhibited a median EQR value (the MarBIT index uses medians rather than averages (Anon. 2006)) slightly below the G/M border of 0.6. EQR values ranged between 0.47 and 0.61. Median EQR values for the southwestern Kiel Bight were 0.50 and slightly below the G/M border. However, values generally ranged between 0.50–0.71, except for deep areas in Eckerförderbucht and Kieler Aussenförde, where values were substantially lower at 0.18 and 0.28, respectively.

Bornholm – western Hanö Bight

EQR values for the western Hanö Bight ranged from 0.24–0.43, with an average value of 0.33. Values were generally above the G/M border. However, the defined acceptable deviation (using BQI) is very high in this area, at 75%.

Mecklenburg Bight

Eastwards from the Kiel Bight, in the Fehmarn Belt area, the median EQR value was 0.62 and ranged between 0.57–0.71. Similar EQR values were found in the Lübeck and the Mecklenburg Bight, with a median EQR of 0.59 and values ranging between 0.55 and 0.64. The G/M border in these areas was set at 0.6.

Southern Baltic Proper

Along the Polish coastline, EQR values were above the G/M border (0.46). An EQR of 0.47 was recorded for the Pomeranian Bay, while an EQR value of 0.67 was obtained for the inner Gulf of Gdansk as well as along the central Polish coast.

Eastern Baltic Proper

In Lithuanian coastal waters, EQR values averaged 0.69 and were generally below the G/M border (0.70–0.83). However, values were highly variable between different regions; the Curonian Lagoon showed an EQR of 0.84, while the northern Lithuanian coastal waters had an EQR of 0.74 and the southern coastal waters had an EQR of only 0.47. Northwards, towards the Gulf of Riga and Pärnu Bay, benthic invertebrate status was classified as good, i.e. EQR values ranged between 0.50 and 0.80. Also in the western Estonian archipelago region and in the Estonian northern coastal waters, a good status was generally obtained.

Western Baltic Proper

West of Öland in the Kalmarsund area, the average EQR value was 0.31, which is slightly above the G/M border of 0.29 (**Table 2.1**). In northern Gotland coastal waters, the mean EQR value was 0.35 (G/M = 0.33), while in the coastal waters west of Gotland, EQR values averaged 0.26 (and were generally below the G/M border), ranging between 0.08 and 0.49. In the coastal mainland waters of the western Baltic Proper, EQR values averaged 0.31 for the entire region. However, clear differences were found between inner and outer coastal areas in this region; in the inner coastal waters EQR values ranged between 0.14 and 0.27, which is below the G/M border, whereas in the outer coastal waters values ranged between 0.24 and 0.58 and were generally above the G/M border.

Archipelago Sea and the western Gulf of Finland

The coastal waters of the Archipelago Sea and the western Gulf of Finland are characterized by large spatial differences in habitat quality and complexity owing to the mosaic nature of this archipelago area. The assessment values obtained using the BBI index (Brackish-water Benthic Index; Perus et al. 2007) illustrated large variations in EQR values, which ranged between 0.00 and 1.02. EQR values averaged 0.54 for both inner and outer coastal waters, which generally were above the G/M border (**Table 2.1**).

Gulf of Finland

In the southern Gulf of Finland in Estonian coastal waters, a good benthic invertebrate status was obtained. EQR values ranged between 0.50 and 0.80. In the northern parts of the Gulf of Finland, along the southeastern Finnish coast, conditions were poorer; here EQR values averaged 0.29, ranging between 0.02 and 0.62, and were generally below the G/M border.

The Åland Sea

Only data from the western part of the Åland Sea were obtained. EQR values ranged between 0.11 and 0.36, with an average of 0.20, while the G/M border varies from 0.29–0.31.

The Bothnian Sea

Along the eastern Finnish coastal waters of the Bothnian Sea, EQR values ranged between 0.44 and 0.76, and averaged 0.65 (G/M = 0.56). Along the western Swedish coast, EQR values for the entire region averaged 0.26, ranging between 0.04 and 0.47. EQR values were generally below the G/M border (G/M = 0.29–0.31). In the southwestern, middle and northwestern Swedish coastal waters, regional EQR values were 0.25, 0.27 and 0.23, respectively.

The Quark

Conditions in the Quark were highly variable and different between eastern (Finnish) and western (Swedish) coastal waters. In the eastern coastal waters, EQR values ranged between 0.29 and 0.86 and averaged 0.62 (G/M = 0.57–0.58). In contrast, in the western coastal waters, EQR values ranged from 0.11 to 0.28 and averaged 0.18 (although here acceptable deviations up to 86% were allowed). The G/M border was set at 0.14–0.31, and western coastal waters were generally below this border.

Bothnian Bay

Along the eastern Bothnian Bay EQR values averaged 0.69, ranged between 0.42 and 1.07, and were mostly above the G/M border (G/M = 0.56–0.58). In contrast, Swedish waters along the western Bothnian Bay showed very different results. Along this coast EQR values ranged between 0.02 and 0.31, with an average of 0.13. The G/M border varied between 0.14 and 0.30 and, hence, the EQR values were generally below the G/M border.

2.7.2. Temporal trends

When examining long-term trends in data collected between 1965 and 2006, it becomes immediately obvious that conditions were already disturbed in the mid-1960s. Benthic invertebrate status in the central parts of the Baltic Sea, in particular, is more or less entirely controlled by the presence or absence of hypoxia/anoxia. Already in Hessle’s (1924) seminal work, hypoxia/anoxia was reported in both coastal and open-sea areas. However, he also reported on the presence of species, such as the polychaete *Scoloplos armiger* at over 140 m depth in the Eastern Gotland Basin, which indicated the limited spatial extent of hypoxic bottom waters at that time. Current evidence suggests that the spatial and temporal extent of oxygen deficiency has increased over the past decades. In the light of historical work (Hessle 1924), it is also likely that reference conditions defined for open-sea areas in this assessment are underestimates. Generally, Baltic benthic macrofauna are characterized by small shallow-dwelling species owing to low salinity and transient hypoxia; historically it was only in the southern Baltic where more mature communities composed of deeper-dwelling, larger species, e.g. some long-lived bivalves and large polychaetes, could have developed (Tulkki 1965, Rumohr et al. 1996). However, currently macrobenthic communities are severely degraded and below a 40-year average in the entire Baltic Sea (Norkko et al. 2007).

Seasonal hypoxia, owing to increased nutrient inputs, has caused mortalities in the benthic communities in the Kattegat since the 1980s. The effects of hypoxia have been very patchy in both space and time, however, and cannot fully explain the general abundance pattern with high densities in the mid-1990s and relatively low values in the assessment period (**Fig. 2.34**). Nor can hypoxia fully explain the long-term decrease in alpha species richness which occurred from the mid-1990s until 2006. The more wide-ranging implications of reduced benthic communities, hypoxia-induced or not, has been observed for demersal fisheries in this sea area (Karlson

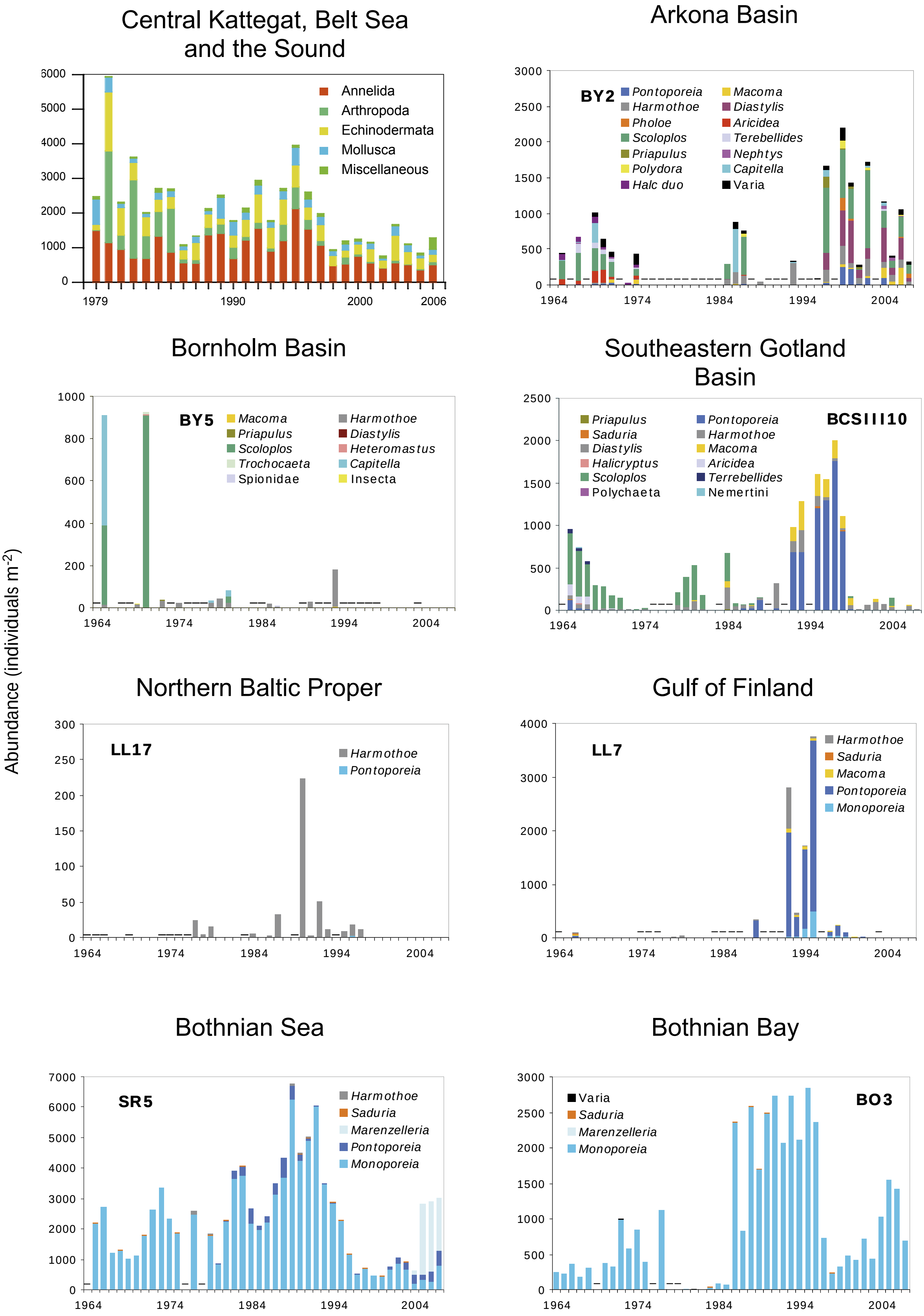


Figure 2.34 Examples of long-term changes in macrobenthic communities in the open-sea areas of the Baltic Sea (Norkko & Jaale (2008), with inclusion of data from the Kattegat (Josefson, unpublished)). Note differences in x-axes and groupings of taxa between the Kattegat and the other sea areas.

et al. 2002). In comparison with the Kattegat, benthic diversity is much reduced in the deep waters of the Arkona Basin, the Bornholm Basin and the southern Gotland Basin, owing to the lower salinity regime (**Fig. 2.34**, Norkko & Jaale 2008). Benthic community composition in this area covaries strongly with oxygen-rich saltwater inflows from the Danish Straits. A shift towards more polychaete-dominated communities, which are more tolerant to eutrophication, has been observed in the Bornholm and Arkona Basins (Karlson et al. 2002). It is often the polychaete *Bylgides (Harmothoe) sarsi* together with *Scoloplos armiger* that dominate the community at deep-water stations, while the bivalve *Macoma balthica* and the amphipod *Pontoporeia femorata* occur only when oxygen conditions improve. Azoic conditions occur repeatedly below the halocline. The southern Baltic has experienced an overall reduction in salinity during the 20th century, which has resulted in a replacement of marine species with brackish-water species (BCSIII-10 in **Fig. 2.34**, Norkko & Jaale 2008). This also highlights the problem of setting reference conditions as baselines shift.

The benthic communities in the northern Baltic Proper and the northern and central parts of the Eastern Gotland Basin are seriously reduced (**Fig. 2.34**). Owing to a permanent halocline and reduced oxygen conditions, this area had impoverished macrozoobenthic communities or azoic sediment conditions during the 1970s and 1980s. Intermittently recovering communities were recorded in the 1990s. The saltwater inflow in 1993 strengthened the halocline, resulting in a lack of zoobenthic communities on approximately one-third of this sea area (Laine et al. 1997; Norkko et al. 2007). *Bylgides sarsi* is the most frequently occurring species at these deep stations (80–170 m), occasionally together with the bivalve *Macoma balthica*, the amphipods *Pontoporeia femorata* and *Monoporeia affinis*, the isopod *Saduria entomon* and the priapulid *Halicryptus spinulosus*. The response of opportunistic benthic species to improved oxygen conditions can be rapid, but with a delay in the recovery of total community abundance and biomass.

In the Gulf of Finland, generally low benthic community abundance, biomass and diversity were recorded below the halocline during the 1960s and 1970s. When the halocline weakened and disappeared because of the prolonged stagnation period from 1977–1993, this resulted in an increased oxygen content of the bottom waters and recovery of the macrozoobenthic communities (Laine et al. 2007). The halocline was re-established in 1993–1994 and the abundant macrobenthic communities recorded in the early 1990s in the deep central parts of the Gulf crashed almost completely in 1996–1997, and have not recovered to any larger extent owing to continued poor oxygen conditions (Norkko et al. 2007). As the oxygen content of bottom waters is reduced, key species in the Gulf of Finland such as *Monoporeia affinis* and *Pontoporeia femorata* disappear, along with more resistant species such as *Macoma balthica* and *Saduria entomon*. The polychaete *Bylgides sarsi* is a fast colonizer in intermittently recovering areas (**Fig. 2.34**).

In the Gulf of Bothnia, low salinity strongly reduces faunal diversity but also prevents the formation of water column stratification and hence makes conditions less susceptible to oxygen deficiency. However, in recent years some low oxygen levels (<40%) have been recorded, possibly due to early-stage eutrophication. Historically, macrobenthic communities have been entirely dominated by the amphipod *Monoporeia affinis*, which exhibits strong natural fluctuations in population abundance and usually comprises 70–100% of total community abundance. Abundances have been severely reduced since the peaks in abundance and biomass in the early to mid-1990s and are generally below the long-term average (Norkko et al. 2007). The reasons for this decline are unknown. However, some recovery has been observed in certain areas of the Bothnian Sea during the past years. The invasive polychaete *Marenzelleria* sp. has spread rapidly throughout most of the Gulf of Bothnia. In the southern Bothnian Sea (station SR5), its abundances increased noticeably between 2004 and 2006 (when it comprised about 80% of total community abundance), but now polychaete numbers appear to be declining and the amphipods *Monoporeia affinis* and *Pontoporeia femorata* are recovering, at least in some areas of the Gulf (**Fig. 2.34**).

3 WHAT ARE THE SOURCES AND LOADS?

The eutrophication signals and the current status, as described in **Chapter 2**, are the result of nutrient enrichment caused by inputs of nutrients from land and air to the sea. This chapter documents both nutrient loads and their sources in the Baltic Sea area. Nutrients originate from a variety of human activities and ultimately arrive in the sea via (1) emissions to air and subsequent deposition, (2) discharges from point sources, and (3) losses from diffuse sources. In addition, nutrients from natural background sources contribute to the load. The emissions, discharges, and losses are transported via air or water to the Baltic Sea, cf. **Fig. 3.1**.

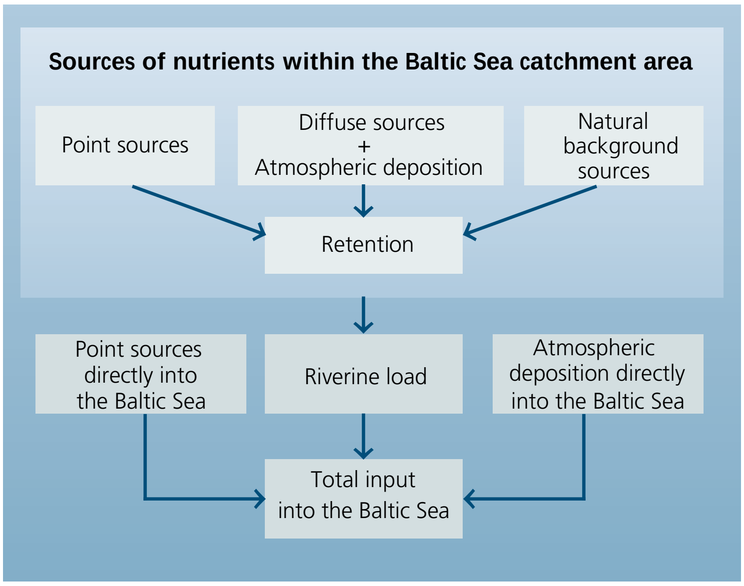


Figure 3.1 Conceptual model of nutrient sources and loads to the Baltic Sea.

Once the nutrients have been deposited, discharged, or lost to the marine environment, they contribute to the nutrient pool in the seawater. Natural internal processes, such as nitrogen fixation and denitrification, affect the nutrient pool as well as the exchange of water between the Baltic Sea and the Skagerrak and North Sea.

3.1 What are the sources and what are the emissions, discharges and losses?

Nutrients enter the Baltic Sea via rivers, as direct discharges from sources located along the coastline, and via atmospheric deposition. Nutrients in riverine discharges originate from the catchment area. They may originate as discharges from point sources, such as industrial or municipal wastewater plants, as losses from diffuse sources, mainly agriculture and scattered dwellings, or as airborne deposition onto the land and waterbodies of the catchment. Natural background sources refer mainly to natural erosion and leakage from unmanaged areas that would occur irrespective of human activities.

The airborne loads in this report refer only to direct atmospheric deposition into the Baltic Sea. They originate from emissions both inside and outside the catchment area of the Baltic Sea.

Another cause of increased nutrient levels in the sea, especially in the case of phosphorus, is the ‘internal load’: phosphorus reserves accumulated in the sediments of the seabed are released back to the water column under anoxic conditions. The magnitude of the nitrogen pool is regulated by nitrogen fixation by cyanobacteria and removal of nitrogen by bacteria-driven denitrification or to a lesser extent by anammox (anaerobic ammonium oxidation).

3.1.1 Emissions to air

The emissions to air both from HELCOM Contracting States as well as from countries outside contribute to the nitrogen load in the Baltic Sea. In addition, emissions from international ship traffic are a significant source of nitrogen deposition.

Anthropogenic emissions have been officially reported to the Convention on Long-Range Transboundary Air Pollution under the UN Economic Commission for Europe (ECE) by HELCOM Contracting Parties and by other members of the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmissions of Air Pollutants in Europe (EMEP). Annual total emissions of nitrogen oxides and ammonia cover the time period 1980–2006 for some of the countries, while for others data only for recent years are available. The geographic locations of the emission points are included in the reporting. In part of the annual reporting, the emissions have been divided into eleven emission sectors specified in the EMEP-CORINAR Emission Inventory Guidebook (European Environment Agency 2007). Expert estimates of the emissions have been calculated to complete gaps or uncertainties in order to be able to make homogeneous model calculations (EMEP WebDab

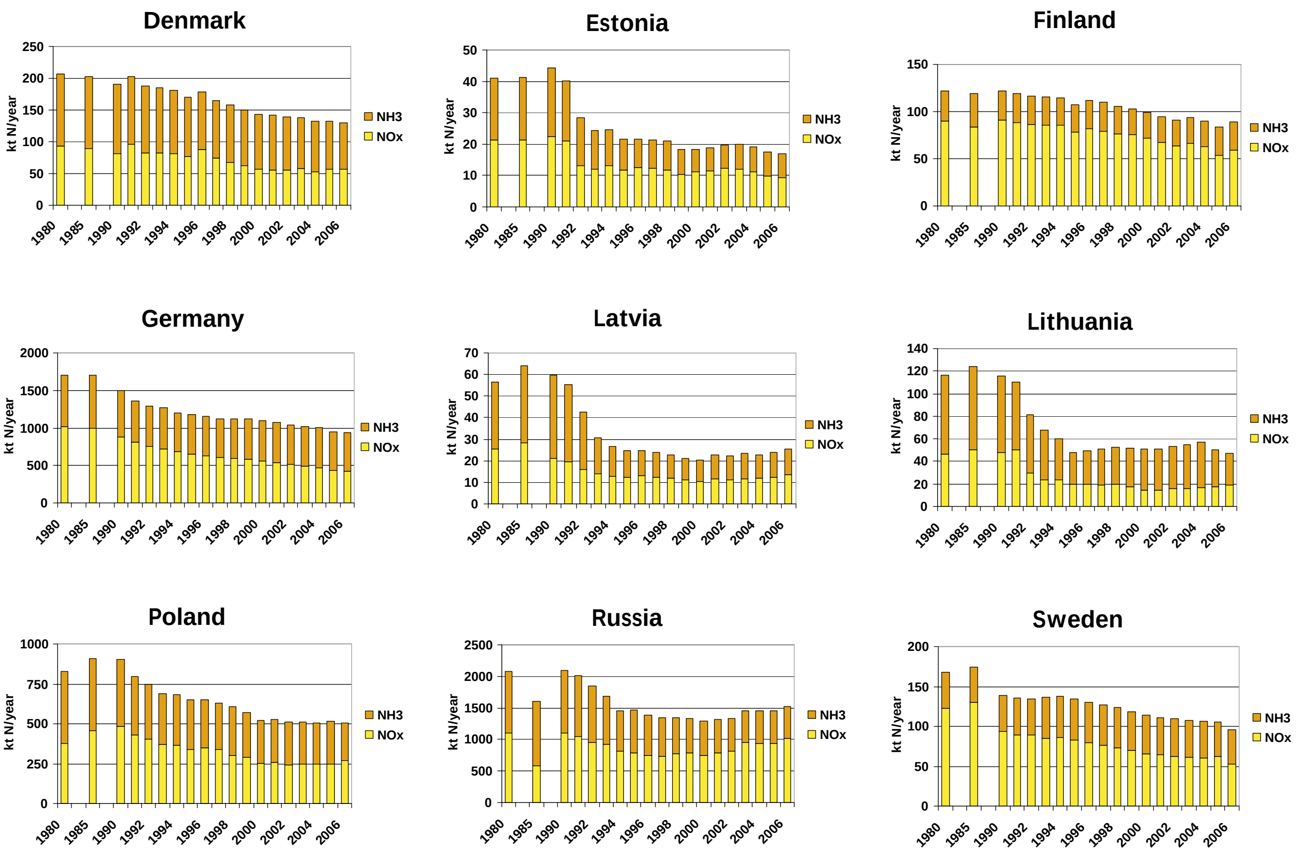


Figure 3.2 Annual atmospheric emissions of nitrogen oxides (NOx), ammonia (NH_3) and total nitrogen in thousands of tonnes (kt) from individual HELCOM Contracting Parties in the period 1980–2006. Data from the EMEP emissions database WebDab, with official emissions corrected using expert estimates (EMEP WebDab 2008). Note the different scales for different countries. Russia refers to the EMEP area.

2008). The largest emissions of nitrogen oxides (NOx) from HELCOM countries arise from sectors such as road transport, other mobile sources, and combustion in energy and transport industries and manufacturing industries. For ammonia (NH_3), the largest emission sector is agriculture with a ≥90% share (Bartnicki et al. 2007). **Fig. 3.2** shows the annual atmospheric emissions of nitrogen oxides, ammonia, and total inorganic nitrogen as the sum of these two from individual HELCOM Contracting Parties in the period 1980–2006 (EMEP WebDab 2008). For most of the countries, a decline has taken place during this period. Since 1980, there has been a reduction of approximately 38% in total nitrogen emissions in air (HELCOM 2007a, Bartnicki et al. 2007).

Fig. 3.3 shows the annual emissions of nitrogen oxides and ammonia in 2004. Heavy emission areas are located in the southwest and south of the Baltic Sea, on the route of the prevailing air transport to the sea.

Shipping is the most important source of nitrogen deposition in the Baltic Sea (Bartnicki et al. 2007). Ship emissions in the Baltic Sea were estimated to be 104,000 tonnes nitrogen in 2005 and 105,000 t in 2006 in the EMEP centre's reports to HELCOM (Bartnicki et al. 2007, Bartnicki et al. 2008). The ShipNodeff-project made a preliminary calculation of the emissions based on observed shipping information in the Baltic Sea. Their estimate is 370,000 t NOx corresponding to 113,000 t of nitrogen annually (Stipa et al. 2007). Recent estimates show that nitrogen oxide emissions from the international shipping traffic on European seas increased by more than 28% between 1990 and 2000 (HELCOM 2007a, EEB 2004). Emissions from shipping are estimated to increase annually by 2–3% (Bartnicki et al. 2007).

In 2008, the International Maritime Organization (IMO) Marine Environment Protection Committee (MEPC) adopted the revised MARPOL Annex VI and the associated NOx Technical Code, both of which will enter into force on 1 July 2010 through a tacit

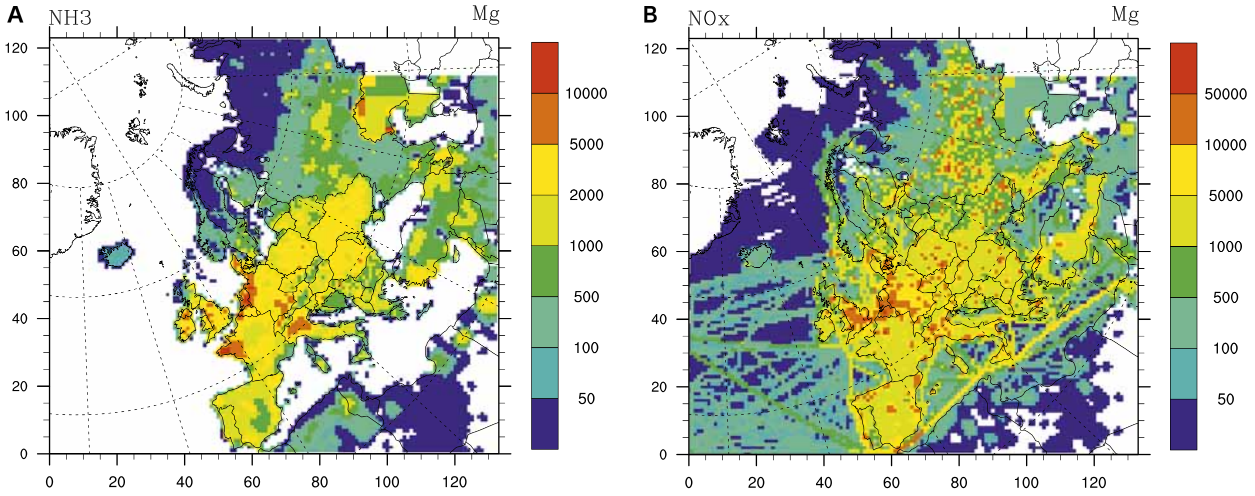


Figure 3.3 Map of ammonia (NH_3) emissions in tonnes (Mg) in 2004 in the EMEP area (panel A) and nitrogen oxide (NOx) emissions (panel B). Official emissions corrected using expert estimates. (EMEP WebDab, http://webdab.emep.int/).

amendment procedure. The main changes brought by MARPOL Annex VI are a progressive reduction in NOx emissions from marine diesel engines and the introduction of Emission Control Areas (ECA) for particulate matter as well as NOx (NECA), in addition to the existing SOx Emission Control Area (SECA) regime. NOx emissions from certain types of ships shall be reduced by 15% compared to the current levels, starting from 1 January 2011, followed by an 80% reduction in NECAs starting from 1 January 2016.

Within the Gothenburg Protocol to the UNECE Convention on Long-Range Transboundary Air Pollution (UNECE 1999) and the EC National Emission Ceilings (NEC) Directive 2001/81/EC, targets have been set for nitrogen emissions. Even if these target values are achieved, the deposition of nitrogen to the Baltic Sea will increase in 2010 compared to the 2003 level mainly because of predicted increases in agricultural activities and shipping (HELCOM 2007a). For example, atmospheric ammonia emissions and ammonium depositions to the Baltic Sea will increase notably if all countries around the Baltic Sea would develop their agriculture to the same level as in Denmark (HELCOM 2007a). The derivation of reduction targets for national atmospheric emissions is based on consideration of critical loads for soil and freshwater systems. Reduction requirements for coastal and marine ecosystems are not taken into account. The Baltic Sea Action Plan (BSAP) reflects this fact accordingly, with a decision that HELCOM Contracting Parties shall make use of assessments of the effects of airborne nitrogen to the Baltic Sea in the revision of their emissions targets.

A revision of the NEC Directive is under preparation in the European Commission. An Integrated Assessment has recently reported on the results of modelling the cost-effective emissions ceilings for several air pollutants including NOx and NH_3. After the modelled cuts, the environmental objectives of the Thematic Strategy on Air Pollution (based on energy projections that correspond to the recent Climate and Energy Package of the European Commission and the national projections of agricultural activities) would be achieved in 2020 (Amann et al. 2008). For the 27 EU countries, further emissions reduction measures would increase reduction efforts for NOx emissions from 53% in the current policy scenario to 58% compared to 2000. Cuts in emissions of NH_3 would increase from 8% to 22%. The majority of NOx reductions would come from industrial energy combustion, while ammonia reductions would involve action in the agricultural sector (Amann et al. 2008).

The BSAP further reflects the above revisions of emission targets with an agreement that all HELCOM Contracting Parties will aim to include emissions from shipping as well as the achievement of ecological objectives for eutrophication in the revision processes.

3.1.2 Discharges and losses from point and diffuse sources to surface waters within the Baltic Sea catchment

In the Baltic Sea catchment area, the major anthropogenic source of waterborne nitrogen is clearly diffuse inputs. They constitute 71%

of the total load into surface waters within the catchment area. Agriculture alone contributed about 80% of the reported total diffuse load. The largest loads of phosphorus originated from point sources (56%), with municipalities as the main source, constituting 90% of total point source discharges in 2000 (**Fig. 3.4**).

These figures represent anthropogenic and natural origins, and amounts of nitrogen and phosphorus at the sources in the catchment area and the direct point source loads into the Baltic Sea are discussed in **Chapter 3.2.2**.

Status 2001–2006

Load inventories with source apportionment in the catchment are carried out only periodically; annual data are not available. The data concerning loads for the year 2006 from the ongoing Fifth Pollution Load Compilation (PLC-5) Project are not yet available. The data currently available indicate that the general situation will not change much from the situation in 2000 (PLC-4 Project). However, the role of agriculture may be somewhat more significant as a result of increased implementation of nutrient removal measures in the municipal sector.

Temporal trends

The progress in reducing waterborne nutrient discharges from point sources has been rather good, with the 50% reduction target (1988 Ministerial Declaration) for phosphorus achieved by almost all HELCOM countries already in 2000. Further reductions in nutrient discharges from point sources are likely in many countries, resulting from the continued implementation of phosphorus removal measures in municipal wastewater treatment plants (MWWTPs). In particular, the new EU member countries have in recent years undertaken extensive modernization actions for both MWWTPs and industries. In Russia, the situation is improving rapidly and the largest single polluter, the city of St. Petersburg, targets a phosphorus removal efficiency of 90% by the year 2012.

In Denmark, Germany, Finland and Sweden, all municipal effluents are treated in MWWTPs. Nearly all of these plants also use advanced (tertiary) treatment methods with phosphorus removal rates of between 80% and 97%. In the year 2001, only about 55% of households in Poland were connected to a municipal sewage treatment. At the plants, 23% of the wastewater received advanced (tertiary) treatment, and 28% biological (secondary) treatment (European Environment Agency 2005). The prevailing level of municipal wastewater treatment is also still far behind HELCOM Recommendations in Russia.

Contrary to the development for point-source loads, the results from the diffuse sector reflect the fact that measures to reduce nutrient loads from agriculture have fallen short of their aims in many Contracting Parties. However, also for agriculture it can be foreseen that the implementation of load reduction measures will support further reductions in nutrient loading (e.g. BSAP, the EC Nitrates Directive and the EU Water Framework Directive). On the other hand, a possible intensification of agricultural activities in the new EU countries and Russia might without further measures lead to an increase in nutrient loads to the Baltic Sea.

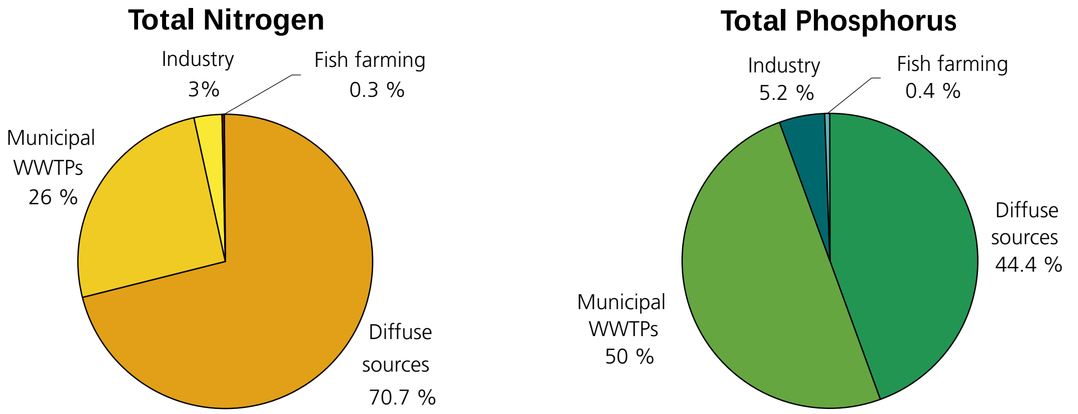


Figure 3.4 Proportion of the inputs of total nitrogen and phosphorus by source into surface waters within the catchment area of the Baltic Sea in 2000 (HELCOM 2004). WWTPs = wastewater treatment plants.

3.2 What are the loads to the marine environment?

The land-based nutrient loads entering the Baltic Sea are either airborne or waterborne. The main pathways of nutrient inputs to the Baltic Sea are: (1) direct atmospheric deposition on the Baltic Sea water surface; (2) riverine inputs of nutrients to the sea, including nutrients that have been discharged or lost to inland surface waters within the Baltic Sea catchment area; and (3) point sources discharging directly to the sea.

3.2.1 Atmospheric deposition

The atmospheric deposition of nitrogen compounds is assumed to comprise about 25% of the total anthropogenic nitrogen load to the Baltic Sea (HELCOM 2005b), while the atmospheric load of phosphorus is only a few percent of the total load to the sea. The proportion of nitrogen from the atmosphere might be larger when the more uncertain parts of the load are better defined. Recent studies of the emissions and load from international shipping, based on real ship traffic, suggest higher emissions and depositions than the previous estimates (Stipa et al. 2007). In addition, the estimates of the atmospheric load of nitrogen to the Baltic Sea mainly ignore organic nitrogen or assume only a minor share of it.

The total atmospheric deposition of nitrogen consisted of slightly more (10-20%) of oxidized (NO_x) than reduced (NH_y) nitrogen (Bartnicki et al. 2008). The amount from wet deposition is much larger than that from dry deposition (Bartnicki et al. 2005). In addition to the inorganic nitrogen load, a more uncertain amount of dissolved organic nitrogen (DON) should be added to the total atmospheric load. Studies from the Bothnian Bay (Rahm et al. 1995) and from oceans (Cornell et al. 1995; Duce et al. 2008) show that the portion of organic nitrogen might be half of the sum of the inorganic nitrogen.

Atmospheric inorganic nitrogen is mostly water soluble and thus ready for use in biomass production. The DON from rainwater has also been shown to stimulate the productivity of coastal marine bacteria and phytoplankton, with 45–75% of DON rapidly utilized by microorganisms (Seitzinger & Sanders 1999).

The quantity of atmospheric nitrogen is studied by monitoring and by mathematical transport model calculations. Monitoring data from coastal and island stations are also used for verification and comparison of the modelled data that cover the whole sea area. The largest uncertainties in the measurements include the representativeness of the stations as well as the sampling. In the model calculations, an uncertainty level of about 30% has been reported (HELCOM 2005b). The largest uncertainties are from emissions inventories, where a typical uncertainty level for the annual total emissions from HELCOM countries was approximately 20% (EMEP 2002). Grid resolutions and simplified descriptions of the complicated air chemistry also add to the uncertainty of the model calculations (Hongisto & Joffre 2005).

Status 2001–2006

The annual flux of atmospheric nitrogen deposition ranges from 300 mg m^{-2} in the northern Gulf of Bothnia to 1,000 mg m^{-2} in the Belt Sea (Bartnicki 2008). The deposition flux of oxidized nitrogen was estimated to be under 200 mg m^{-2} in the Gulf of Bothnia and 200–500 mg m^{-2} elsewhere in 2005 (Bartnicki et al. 2007). The deposition flux of reduced nitrogen was estimated to exceed the level of 200 mg m^{-2} in areas south of Gotland in 2005 (Bartnicki et al. 2007).

The total atmospheric deposition of nitrogen to the entire Baltic Sea basin was estimated at 196,000 t in 2006 in the latest EMEP Centres joint report to HELCOM (Bartnicki et al. 2008), see **Fig. 3.5**. The seasonal variation reflected the variation in the amount of precipitation, with the largest load in October (27,200 t) and the smallest in July (8,300 t) (Bartnicki et al. 2008). The atmospheric nitrogen load is highly episodic. Wet deposition of nitrogen was clearly linked to precipitation over the Kattegat, with large short-term temporal variations, whereas dry deposition of nitrogen was relatively constant (Carstensen et al. 2005). The inter-annual variation in the annual depositions during 2001 and 2006 was from 196,000 t to 224,000 t (Bartnicki 2007a; Bartnicki 2008; HELCOM 2007a). The highest value was estimated in 2001, the lowest in 2002.

Other estimates for the atmospheric nitrogen load have been published for some of the years 2001–2006. Hongisto & Joffre (2005) pre-estimated the

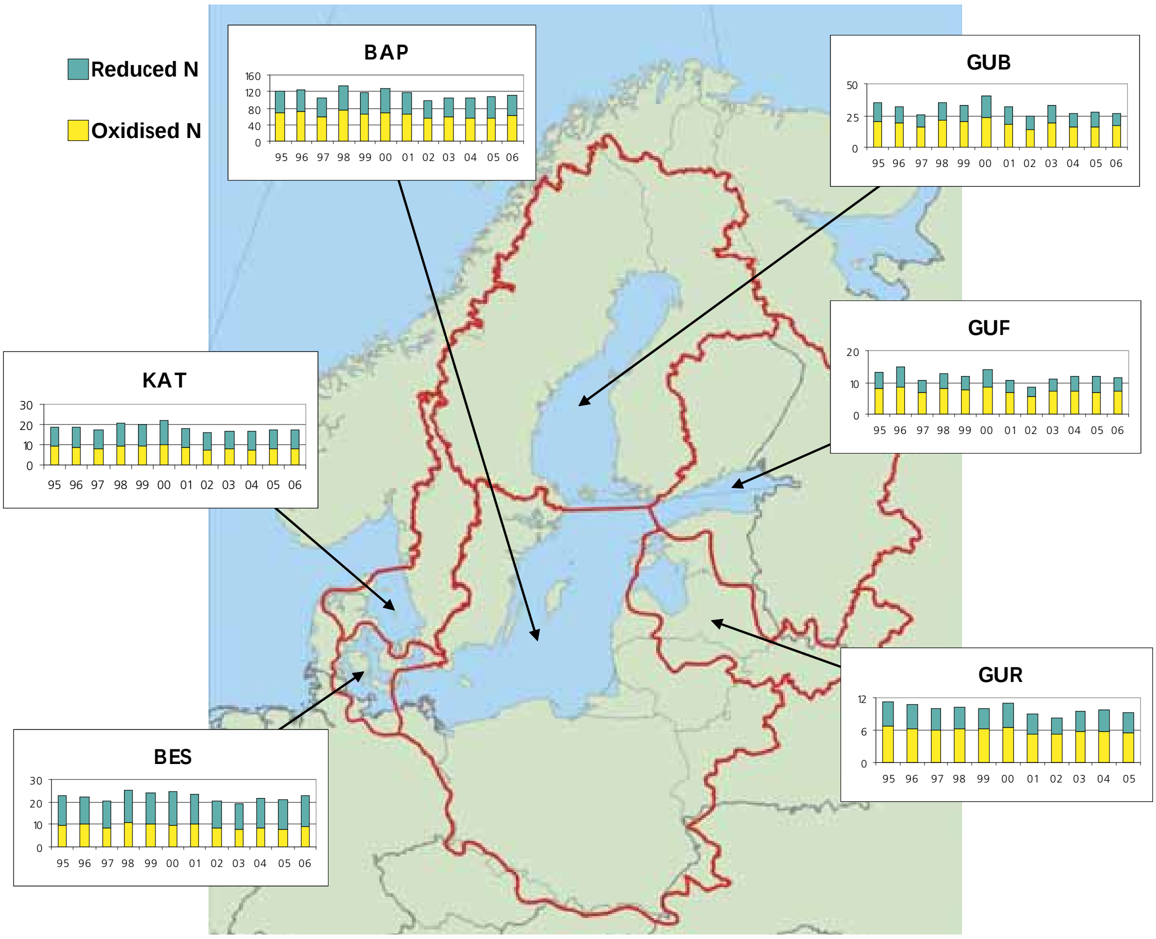


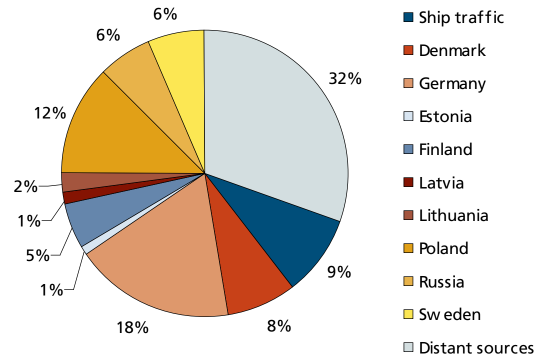
Figure 3.5 Atmospheric deposition of oxidized, reduced and total nitrogen in 1995–2006 to the sub-basins of the Baltic Sea, calculated by the EMEP Meteorological synthesizing centre – West (MSC/W). Units are kt N per year. The scales for the different areas are different (Bartnicki 2008).

total nitrogen deposition at 263,000 t in 2001 and 224,000 t in 2002 using 1998 emissions.

A large load of the atmospheric nitrogen is initially deposited on the soil or vegetation in the drainage area and transported to the sea via rivers after a time lag. This part is counted in the nitrogen load as riverine input. It has been estimated that almost 35% of the total nitrogen load entering the whole Baltic Sea originates from airborne loads (HELCOM 2003b).

The most important source sectors for atmospheric nitrogen emissions are road transportation, fossil fuel combustion in energy production, and shipping for NOx and agriculture for NHy (Bartnicki et al. 2007, HELCOM 2007a), see **Fig. 3.6**. International shipping in the Baltic Sea is estimated to be the most important contributor to the NOx deposition, and third on the list for the total nitrogen deposition (Bartnicki et al. 2007). In addition, shipping on the North Sea contributes about 10,000 t NOx nitrogen to the Baltic Sea. Recent estimates showed that nitrogen oxide emissions from the international shipping traffic increased by more than 28% between 1990 and 2000 (EEB 2004). Calculations using emissions based on real ship traffic estimate that the annual NOx deposition from the Baltic Sea shipping is typically under 20% of the total NOx deposition; however, in July, in small areas in the Northern Baltic Proper, the contribution from shipping emissions can extend up to 50% of the NOx deposition (Stipa et al. 2007).

Among the HELCOM Contracting Parties, Germany and Poland contribute most to the atmospheric nitrogen deposition (**Fig. 3.6**). For the deposition on their neighbouring sub-basins, Finland, Estonia and Denmark were also significant contributors (Bartnicki & van Loon 2005).



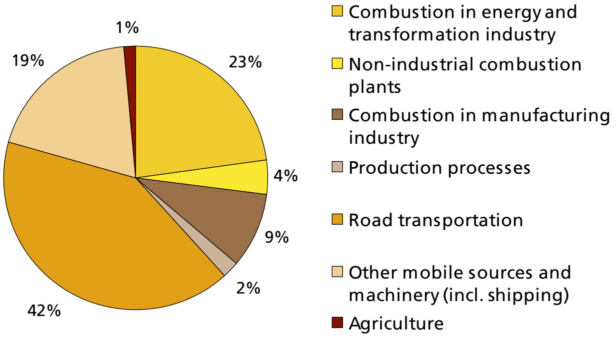


Figure 3.6 Contribution of different emission sources to the atmospheric deposition of nitrogen. Left diagram: percentage of total emissions of nitrogen oxides (NOx) from different sectors in the HELCOM Contracting Parties in 2005 (HELCOM 2007a). Right diagram: Proportion of contribution by source to the atmospheric deposition of nitrogen entering the Baltic Sea basin in 2005; over 30% of the total nitrogen load originates from sources outside the HELCOM area (HELCOM 2007a).

32% of the atmospheric nitrogen is transported from countries outside the drainage area of the Baltic Sea; Great Britain, Ukraine and France are important foreign contributors (Bartnicki et al. 2007; HELCOM 2007a).

Temporal trends

Since 1980 there has been a reduction by approximately 38% in the levels of total nitrogen emissions from HELCOM countries (HELCOM 2004; HELCOM 2007a; Bartnicki et al. 2007). Deposition to the Baltic Sea has declined less, by roughly 33%, during the same period (HELCOM 2007a). This is due to the fact that deposition is strongly dependent on meteorological patterns and the emissions of other countries. The relative contribution from shipping has increased owing to increased shipping emissions and partly because regulations and technical solutions have decreased the nitrogen emissions on land.

EMEP estimated the nitrogen deposition to the Baltic Sea in 2010 for three different emission scenarios within the fulfilment of the targets for nitrogen in the Gothenburg Protocol to the UNECE Convention on Long-Range Transboundary Air Pollution and the EU NEC Directive and/or the levels foreseen to be achieved for 2010. The scenarios were: (1) emission projections according to agreed emission ceilings under the EU NEC Directive and the Gothenburg Protocol and the ENTEC projections for shipping, (2) as for scenario 1, but with 10% lower emissions, and (3) a current legislation (CLE) emission scenario (Bartnicki & van Loon 2005). According to the results, the 2010 depositions will be higher than the 2002 level for all scenarios; NOx depositions will decrease in large parts of the Baltic Sea, whereas the NHy depositions will increase (Bartnicki & van Loon 2005).

Climate change will probably result in increased atmospheric nitrogen deposition in association with increasing precipitation. Growth in the Baltic Sea has been estimated to be 0–20% for the period from 1961–1990 to 2071–2100 (Hole & Engardt 2008). Climate change will also contribute to the atmospheric deposition by changing the pathways of the emissions and affecting deposition processes.

In the future, an increase in agricultural activity in the catchment area of the Baltic Sea and the dominant upwind transport direction will increase the NH_3 emissions and NHy depositions to the Baltic Sea, if strict control measures are not implemented. Over the next 20 to 25 years, the proportion of NH_3 emissions will likely increase owing to enhanced atmospheric emission controls that are predicted to be more effective for NO_x than for NH_3 (Duce et al. 2008). In addition, it is assumed that the international ship traffic in the Baltic Sea will increase at an annual rate of 2–3% (Bartnicki 2007b). A shift to faster ships also tends to increase the emissions.

3.2.2 Riverine and direct loads to the Baltic Sea

Information on the riverine loads of nitrogen and phosphorus is of key importance in following up the long-term changes in nutrient loading and determining the priority order of different sources of nutrients in the eutrophication of the Baltic Sea, as well as for assessing the effect of measures taken to reduce nutrient loading. Quantified load data are a prerequisite to interpret and evaluate the state of the marine environment and related changes in the open sea and coastal waters.

About 75% of the nitrogen and at least 95% of the phosphorus enters the Baltic Sea waterborne (i.e. via rivers or as direct discharges). The atmospheric deposition of nitrogen to the Baltic Sea comprises about one quarter of the total nitrogen load to the Baltic Sea. Phosphorus enters the Baltic Sea mainly as waterborne input, but can also enter as atmospheric deposition; however, as the estimated contribution is only 1–5% of the total phosphorus input, it is not considered in this report.

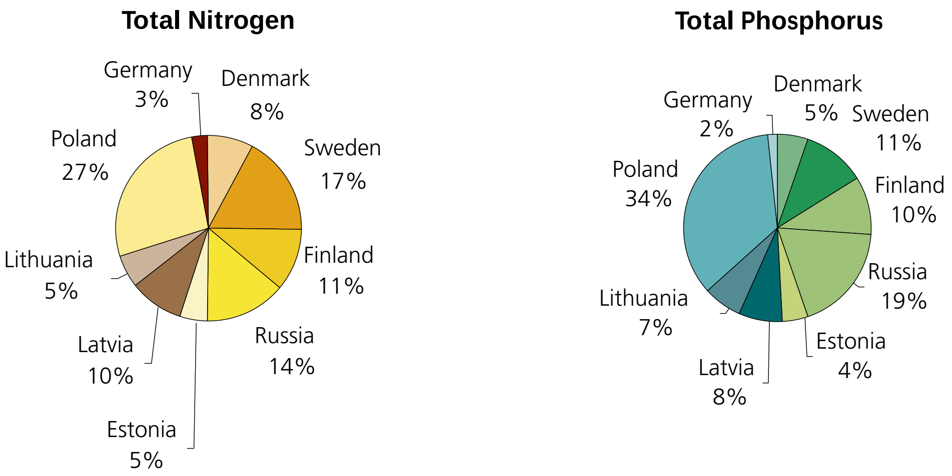
The catchment area of monitored rivers covers 97% of the total Baltic Sea catchment area. Riverine nutrient loads consist of discharges and losses from different sources within a river's catchment area, including discharges from industry, municipal wastewater treatment plants, scattered dwellings, losses from agriculture and managed forests, as well as natural background losses and atmospheric deposition. According to the PLC-4 report, diffuse load (mainly agriculture) contributed almost 60% of waterborne nitrogen loads and 50% of phosphorus loads.

Status 2001–2006

In 2001–2006, the average annual total waterborne (riverine, coastal areas, and direct point and diffuse sources) load of nitrogen entering the Baltic Sea amounted to 641,000 t, and total phosphorus 30,200 t. The average annual runoff in 2001–2006 was 14,200 $m^3\ s^{-1}$. The proportions of average annual nitrogen and phosphorus inputs into the Baltic Sea by HELCOM countries in this period are presented in **Fig. 3.7.** The main contributors of nitrogen were Poland (27%), Sweden (17%), and Russia (14%). The largest loads of phosphorus originated from Poland (34%), Russia (19%), and Sweden (11%).

Temporal trends

Compared with the previous six-year period (1995–2000), total loads decreased for both nitrogen (Russian load estimated in 1995-1999 and 2005) (−13.7%) and phosphorus (−15.3%), see **Figs. 3.8** and **3.9**. At the same time, the average annual flow also decreased by 9.8%. Therefore, it is obvious that almost two-thirds of the observed decrease can be explained by the differences in hydrological conditions during these two periods. Nutrient fluxes vary considerably from year to year depending mainly on hydrological conditions. In periods of high runoff, nutrients are abundantly leached from soil, thus increasing the loads originating from diffuse sources and natural leaching.



Total Nitrogen

Total Phosphorus

Figure 3.7 The average annual proportions of total nitrogen (left diagram) and total phosphorus (right diagram) inputs into the Baltic Sea by HELCOM countries in the period 2001–2006.

Figure 3.8 Riverine and direct point source inputs into major Baltic Sea basins 1994–2006. Note the different scales for the different areas. P fractions: light green (lower part of the column) = PO_4-P, light+dark green = Tot-P. N fractions: lowest part (yellow-orange) = NH_4-N; mid-section (bright yellow) = NO_2+NO_3-N; upmost section (light yellow) = organic N, full bars = Tot-N. Russian data on Tot-N load into the Gulf of Finland and Baltic Proper is missing in 1994-1999 and in 2005.

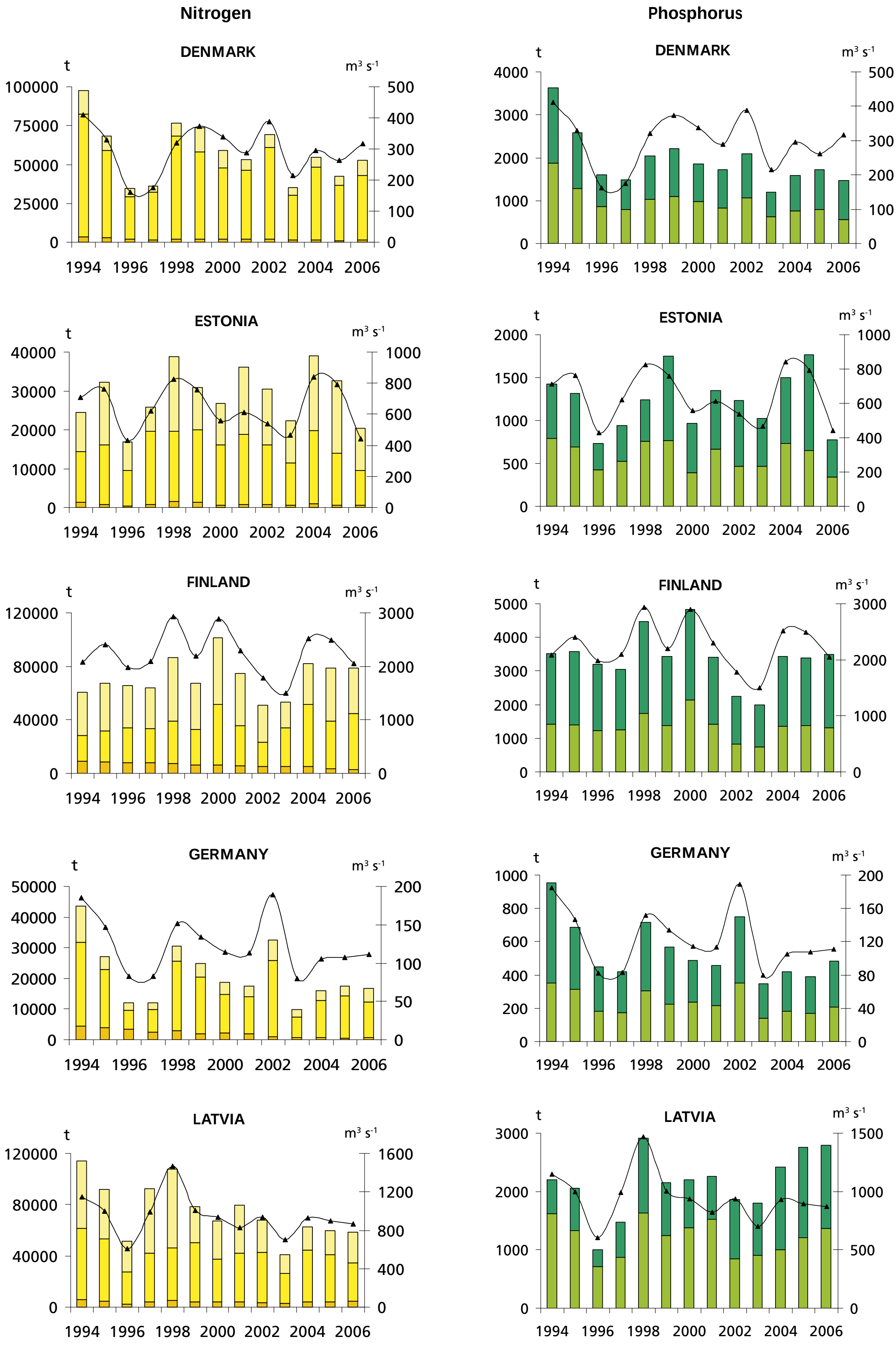


Figure 3.9 Riverine and direct point source loads from Baltic Sea countries, 1994–2006. Note the different scales for the different countries. P fractions: light green (lower part of the column) = PO_4-P, light+dark green = Tot-P. N fractions: lowest part (yellow-orange) = NH_4-N; mid-section (bright yellow) = NO_2+NO_3-N; upmost section (light yellow) = organic N, full bars = Tot-N. Continued in the next page.

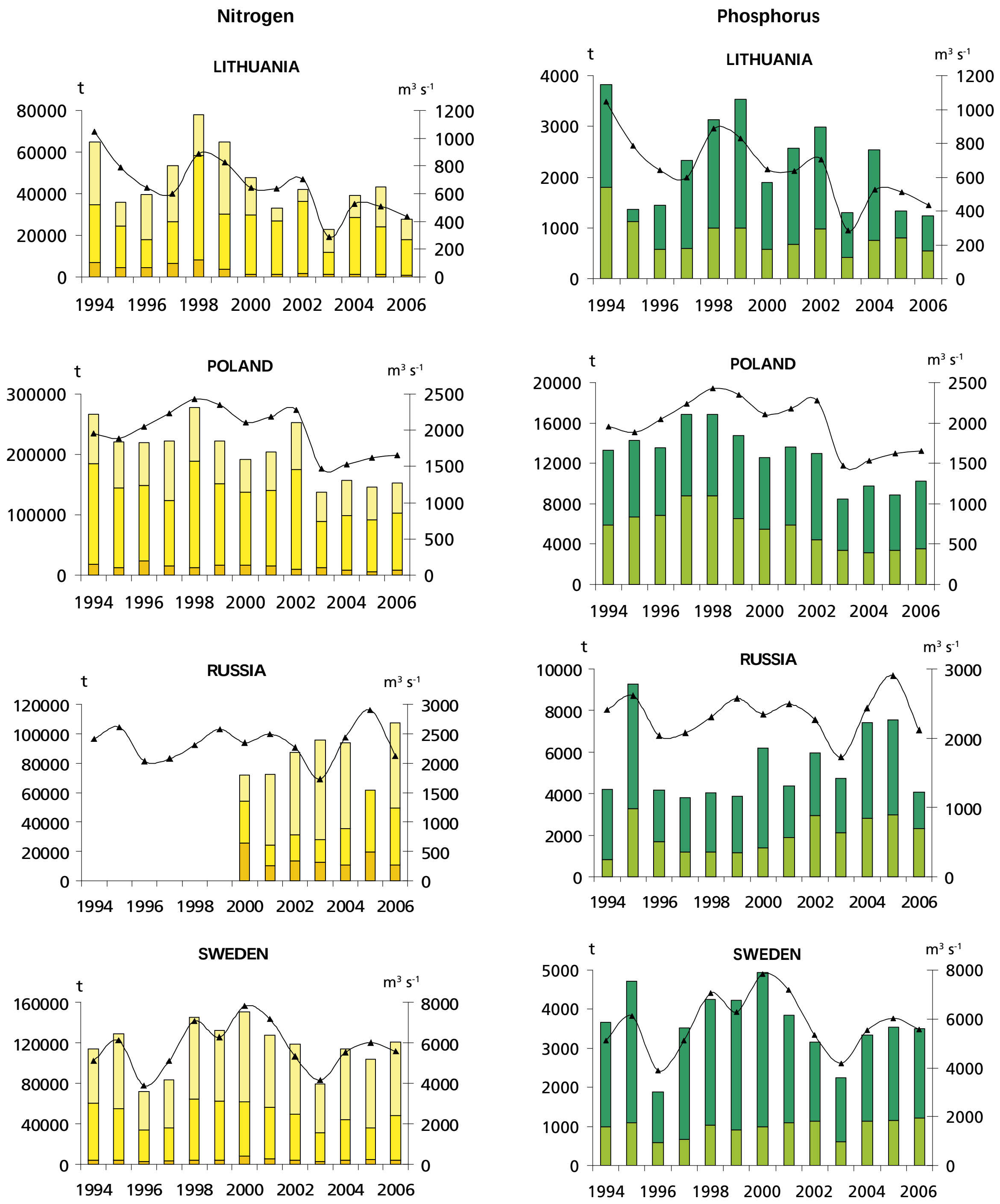


Figure 3.9 Continued from page 79. Riverine and direct point source loads from Baltic Sea countries, 1994–2006. Note the different scales for the different countries. P fractions: light green (lower part of the column) = PO_4-P, light+dark green = Tot-P. N fractions: lowest part (yellow-orange) = NH_4-N; mid-section (bright yellow) = NO_2+NO_3-N; upmost section (light yellow) = organic N, full bars = Tot-N. Russia did not report Tot-N in 1994-1999 and in 2005.

Riverine nutrient discharges, especially of phosphorus, appear to have decreased during the entire 13-year period from 1994–2006 for which annual data are available from the Contracting Parties. In addition to the hydrological changes, this most probably also reflects the implementation of load reduction measures in the catchment area (mainly improved treatment of municipal and industrial wastewaters). It is also known that the load reduction measures are particularly efficient for phosphorus in municipal wastewater treatments plants, which is reflected in the larger decrease in phosphorus load (**Fig. 3.10**). This conclusion is also supported by the data on direct discharges of P and N

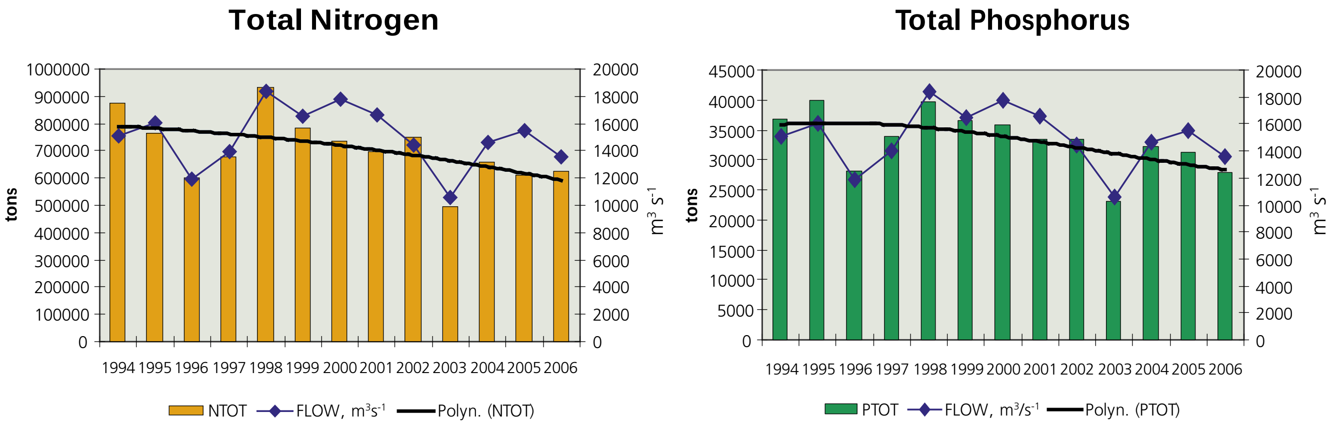


Figure 3.10 Riverine and direct point-source inputs into the Baltic Sea from 1994–2006. NTOT = Total nitrogen; PTOT = Total phosphorus.

into the Baltic Sea during the periods mentioned above. Comparable data for these periods are available from five Contracting Parties (Germany, Denmark, Finland, Lithuania, and Poland). Direct discharges decreased between these two periods in all cases for both P and N. The average decrease varied from 16.4% in Poland to 66.5% in Germany for nitrogen and from 23.4% in Finland to 68.9% in Lithuania for phosphorus. The proportion of direct point-source discharges in the total load was significant in Denmark and Finland. In Denmark, the decrease was around 40% for both P and N and in Finland it was around 25% respectively.

The largest area-specific nitrogen inputs into the Baltic Sea occurred in the catchment areas of the Danish Straits (1014 kg N km^{-2}) and the Gulf of Riga (839 kg N km^{-2}). The highest area-specific phosphorus loads occured in the catchment areas of the Gulf of Riga (36,9 kg P km^{-2}) and the Baltic Proper (26,2 kg P km^{-2}). In contrast to these findings, the area-specific nitrogen and phosphorus loads from the Gulf of Bothnia (216 kg N km^{-2}, 8.9 kg P km^{-2}) and the Gulf of Finland (390 kg N km^{-2}, 12,7 kg P km^{-2}) catchments – both with extensive pristine areas – were considerably lower.

3.2.3 Inputs from adjacent seas

Ærtebjerg et al. (2003) presented a nitrogen nutrient budget for the Kattegat/Belt Sea area including transports between the Kattegat and the Skagerrak. The budget was based on gross advective transports from the Baltic Sea and the Skagerrak in the period 1974–1999. The bioavailability of the nitrogen in the different sources was calculated from measured concentrations of inorganic nitrogen and nitrogen incorporated into phytoplankton, and compared to experimental results (Kaas et al. 1994). Including the bioavailability of the nitrogen sources in the budget increased the Danish contribution to 25% of the gross supply to the Kattegat/Belt Sea area. However, some of the nitrogen supplied from the Skagerrak actually originates from the Kattegat and some is removed by denitrification or exported to the Baltic Sea. Taking this into account increases the Danish contribution of bioavailable nitrogen to 32%, the direct contributions from Sweden and Germany to 11% each, the contributions from the Baltic Sea and the Skagerrak to 14% and 19%, respectively, and the contributions from other European countries via the atmosphere to 13%.

3.2.4 Sediment release of phosphorus

The sediment release of nutrients, often termed ‘internal loading’, affects the balance of phosphorus in the Baltic Sea, especially in the Baltic Proper, the Gulf of Finland and the Gulf of Riga. The term ‘internal loading’ as such often causes misinterpretations. Intra-annually, the main part of both sediment nutrient accumulation and release is based on the production of autochthonous organic matter sedimenting during the same year, in particular after the vernal plankton bloom. Thus, within one and the same year, nutrients are first sedimented and then partly released.

On a basin scale, tentative estimates of net P release from sediments can be made as the difference between the nutrient pools of two succeeding winters. Conley et al. (2002a) studied the role of internal biogeochemistry for the pool of inorganic P in the Baltic Proper and the Gulfs of Finland and Riga by using extensive monitoring data from 1970 to 2000. According to their calculations, the largest single net increase of the P pool (indicating sediment release) was estimated as 90,000 t yr^{-1}, while the largest annual net decrease (indicating sediment binding) was about 110,000 t yr^{-1}. Both values are much larger than the external annual total P load and its variation, given as 23,000 to 37,000 t yr^{-1} into the basins studied by Conley et al. (2002a).

Over longer time periods, the average annual changes in the pool of P (assumed as net annual exchange of P between sediment and water) are small and depend strongly on the hydrodynamic conditions of the chosen period. Both the study of Conley et al. (2002a) and the long-term trends presented by Fleming-Lehtinen et al. (2008) and in **Chapter 2** suggest that after the mid-1980s more a decreasing than an increasing trend was evident in average winter P concentrations in the Baltic Proper, and thus in most years the average net flux of P evidently has been from water to sediment. However, taking into account the large bottom area of the Baltic Proper, its sediment P retention capacity is relatively low, and it is not capable of retaining its external P load; thus, it acts as a net source of P for the neighbouring basins (see **Tables 3.3** and **3.4**).

The Gulf of Finland serves an example of a Baltic Sea sub-basin that is occasionally strongly affected by increased sediment P release (**Fig 3.12**). In the 1990s, inorganic P concentrations in the Gulf increased despite a decrease of 34% (about 3,000 t yr^{-1}) in external P loading (Pitkänen et al. 2001). The experimentally measured sediment-water P fluxes from 27 anoxic sites all over the Gulf suggested an average release of 13 mg m^{-2} d^{-1} (Lehtoranta 2003). It was calculated that the sediment release could

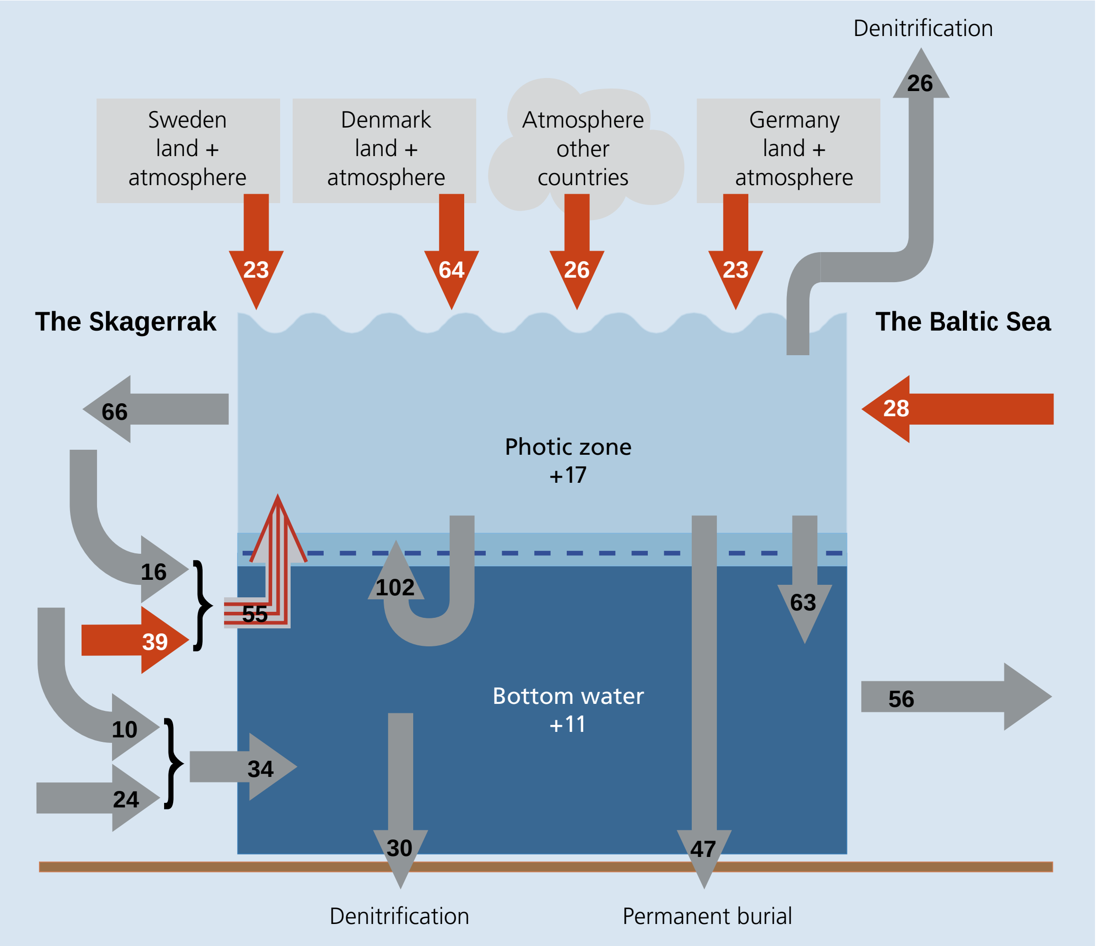


Figure 3.11 Transport of biologically active nitrogen in the transition zone between the Skagerrak and the Baltic Sea. The values in the diagram are tonnes N per year. From Ærtebjerg et al. (2003).

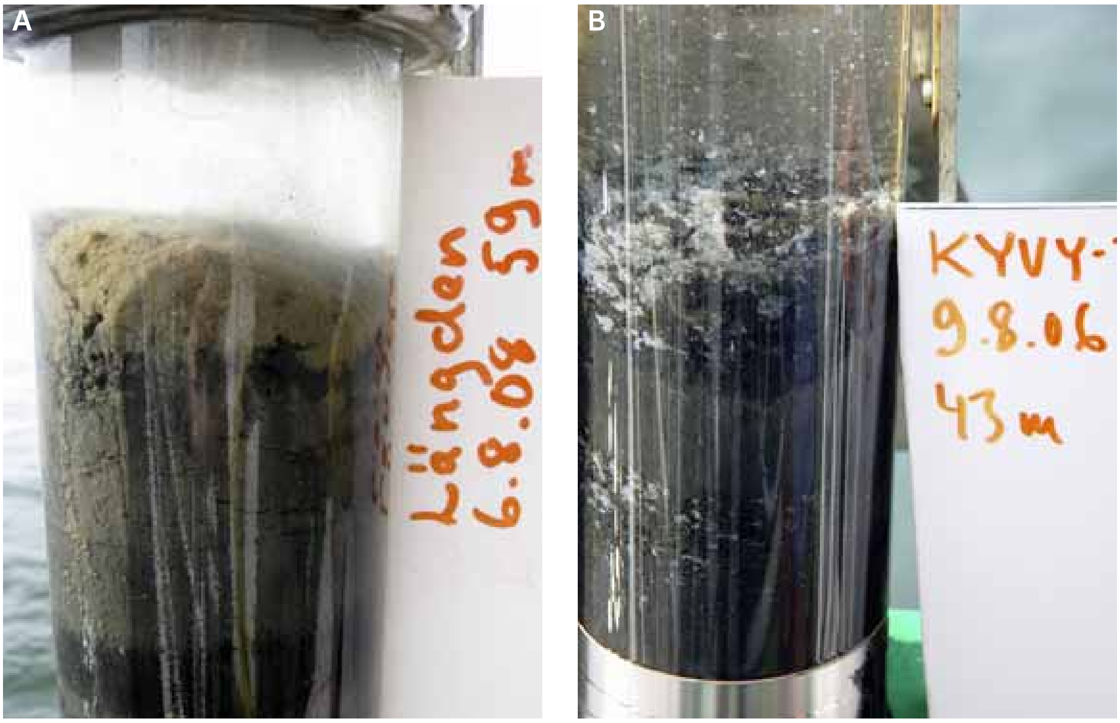


Figure 3.12 Healty, oxidized (panel A) and reduced (panel B) bottom sediments in the Gulf of Finland in 2008 and 2006, respectively.

largely explain the estimated annual increase in the content of inorganic P in the Gulf of about 10,000 t yr^{1} between the winters of 2001 and 2002 (Pitkänen et al. 2003). Large increases in inorganic P concentrations also took place in 1996/1997 and 2003/2004 in the Gulf of Finland. In all these cases, salinity stratification strengthened which was rapidly reflected by deep-water hypoxia and reduced conditions at the sediment-water interface leading to increased sediment P release. A part of the increase is probably explained by increased deep-water fluxes of P from the Baltic Proper, especially in the western Gulf of Finland. According to Savchuk (2005), the average influx from the Baltic Proper from 1991–1999 was 1,300 t yr^{-1}. The missing correlation between near-bottom salinity and inorganic P in the eastern Gulf of Finland suggests that there the main origin of the ‘extra’ P is from local sediments (Pitkänen et al. 2001).

The large sediment efflux of P occurs during the years of reduced conditions at the sediment-water interface (Lehtoranta 2003). The reason for the high efflux is large loading and eutrophication (increased sedimentation of organic matter) combined with hydrodynamic properties, i.e., salinity stratification which hampers the regular renewal of near-bottom waters. Anoxic conditions caused by the decomposition of sedimented organic matter and restricted deep-water renewal favour microbial sulphate reduction and iron sulphide formation, which blocks the coupled iron and phosphorus cycling. This, in turn, leads to the release of P because iron is buried as iron sulphides and there are not enough iron oxides in the sediment to bind all the dissolved P. Thus, the microbial processes related to iron cycling in sediments and the trophic state seem to control the release of P in addition to hydrodynamic conditions (Lehtoranta et al. 2008).

It is important to note that the magnitude of sediment P release (‘internal loading’) varies greatly from year to year, largely as a result of physical forcing. Although in some years, especially in the Baltic Proper and the Gulf of Finland, the total annual sediment P release can be considerable and several times larger than the external nutrient load, the long-term average annual net release is small compared with external loads. As a whole, the Baltic Sea retains about 60% of its external P load. Owing to the large intra- and inter-annual variations, far-reaching judgments on the relative roles of different input factors (e.g. external

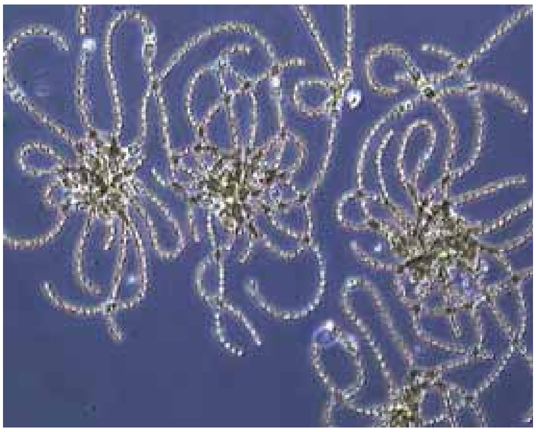


Figure 3.13 *Anabaena lemmermannii*, one of the components of the nitrogen-fixing cyanobacteria blooms in the Baltic Proper.

loading vs. sediment release) cannot be made on the basis of one or a few years alone, despite the clear effect on trophic conditions in those particular years leading, for example, to extensive cyanobacteria blooms.

3.2.5 Nitrogen fixation

Nitrogen fixation is an important source that replenishes the nitrogen lost by sedimentation and denitrification. It is accomplished mainly by filamentous cyanobacteria, in the Baltic Sea especially by the genera *Nodularia* (**Fig. 3.13**), *Aphanizomenon* and *Anabaena*. However, small coccoid cyanobacteria may also significantly contribute to nitrogen fixation (Gallon & Chaplin 1988; Zehr et al. 2001).

Calculations of nitrogen fixation rates in the ocean have been revised substantially upwards during the last decades (Gruber 2005). Early estimates of the annual nitrogen fixation in the central and northern Baltic Proper, including the Gulf of Finland, amounted to 100,000 t N according to Melvasalo et al. (1983) and Leppänen et al. (1988). Rönner (1985) estimated an annual nitrogen input of 130,000 t N by nitrogen fixation for the Baltic Proper and the Gulf of Finland. These early rate measurements may have underestimated nitrogen fixation because they frequently used nets for enrichment and therefore missed a part of the diazotrophs. Wasmund et al. (2001) measured a significant nitrogen fixation in the size fraction <10 µm and claimed that small coccoid cyanobacteria are responsible, whereas Stal et al. (2003) found no nitrogen fixation by picocyanobacteria in the Baltic Sea. These authors also measured significant nitrogen fixation during the night, whereas earlier authors mostly disregarded the activity during the night. By avoiding these sources of errors, Wasmund et al. (2001) measured a mean annual nitrogen fixation of 125 mmol N m^{-2} yr^{-1} in 1997/1998 and estimated a total of 370,000 t N for the Baltic Proper. However, they did not cover the early stages of the developing bloom, which are most active (Ohlendieck et al. 2000). New measurements by Wasmund et al. (2005) in the year 2001 revealed a nitrogen fixation from May to August of 138 mmol N m^{-2}, corresponding to 434,000 t N for the Baltic Proper. It must be emphasized that the apparent increase is due to experimental changes. However, measurements by Degerholm et al. (2008) in 1998–2000 resulted in much lower nitrogen fixation rates, ranging from 56,000 to 125,000 t N in the Baltic Proper. The year-to-year variability may be enormous, as shown by Hübel & Hübel 1995), who calculated an annual nitrogen fixation of 185,950 t N and 18,200 t N for the years 1975 and 1985, respectively, for the Baltic Proper and Mecklenburg Bay. These experimental rate measurements are not always representative because they cover only distinct stations with rather low frequency. Alternative integrating approaches become more important for estimations of nitrogen fixation.

Struck et al. (2004) measured the isotopic ratios of nitrogen in the sediments of the Gotland Basin and found increasing potential summer nitrogen fixation during the eutrophication period from 1969-1973 to 1974-1978, but a strong decrease in the period from 1989-1993 to 1994-1998. The consideration of five-year periods instead of annual data levelled out strong year-to-year fluctuations. The inter-annual variability in nitrogen fixation in the period 1994–1998 at the Landsort Deep station near Stockholm may reach from 61–140 mmol N m^{-2} yr^{-1} according to Larsson et al. (2001). Their approach, based on the increase in total nitrogen except N2 gas, was applied in addition to other methods by Wasmund et al. (2005), who calculated a range of nitrogen fixation from 134–182 mmol N m^{-2} in the Eastern Gotland Basin in 2001. Rolff et al. (2007) also used this method and calculated an annual nitrogen fixation rate of 92 mmol m^{-2} which corresponds to a basin-wide Baltic nitrogen fixation (excluding the Gulf of Bothnia) of 310,000 t N in 2002. Using an alternative method based on measurements of the decrease in dissolved inorganic carbon due to primary production, Schneider et al. (2003) estimated an annual nitrogen fixation in the Baltic Proper of 318 ± 53 mmol m^{-2}. This figure also included a significant nitrogen fixation in the spring, which was not found by other authors but will be a matter of future research.

3.2.6 Sediment nitrogen removal processes

Denitrification is a bacterial process in which nitrate and nitrite are successively reduced to gaseous nitrogen. The process occurs at oxic-anoxic interfaces, mainly in oxygenated surface sediments (Tuominen et al. 1998). Denitrification takes place in estuarine, coastal and deep basin sediments, but is effectively stopped by complete anoxia as the process requires the presence of nitrogen oxides. In anoxic sediments another process, anammox, can reduce NH_4^--nitrogen to gaseous form but is significantly slower. Denitrification removes nitrogen from aquatic ecosystems, including from the Baltic Sea, and partly balances N-enrichment caused by anthropogenic loading and N-fixing (see **Chapter 3.2.5**). Studies of the N removal capacity of sediment denitrification have been carried out in the Gulf of Finland, Gulf of Bothnia, and the Baltic Proper. In the Gulf of Finland, Tuominen et al. (1998) estimated the annual mean nitrogen removal by denitrification to be 45,000 t N yr^{-1}, equalling approximately 30% of the external annual N loading to the Gulf. The estimate by Stockenberg & Johnstone (1997), 31% for the Bothnian Sea and 23% for the Bothnian Bay, is of same order of magnitude. Recent isotopic budget calculations by Voss et al. (2005) suggest that sediment denitrification in the Baltic Proper is 855,000 t N yr^{-1}, corresponding to the amount of cyanobacterial nitrogen fixation. Although sediment denitrification removes nitrogen effectively, it can also be disrupted by eutrophication, as the proportion of removed nitrogen diminishes along with increasing N-concentration (Tuominen et al. 1998 and references therein) and an increase in the area of anoxic sediments can greatly decrease the denitrification capacity of the system, potentially leading to an amplification of N-enrichment.

3.2.7 Advective transports and basin-wise nutrient budgets

Calculating nutrient budgets for the Baltic Sea and its sub-basins is a useful tool to quantify and analyse the role of different external and internal fluxes both within and between sub-basins, for example, for tentative assessments of the most available and cost-efficient protection measures. The basic method to estimate basin-wise budgets and advective transports is to form mass balance equations including all the essential input and output terms with the aid of available data on concentrations, external loading, and internal biogeochemical processes. The water balance between neighbouring 'landward' and 'seaward' basins is usually calculated from Knudsen's formula, based on river inflow and the role of sea salt (salinity) as a conservative tracer (see, e.g., Savchuk 2005).

Nutrient budgets for the Baltic Sea or its sub-basins have been presented by Wulff & Stigebrandt (1989), Perttilä et al. (1995), Wulff et al. (2001b), Savchuk (2005) and Savchuk & Wulff (2007). The data used for these budgets are from different periods, but the basic methodology is the same in all studies, except for Savchuk & Wulff (2007), where also simple biogeochemical modelling has been used to help to form the budgets. Additionally, in that study, the Baltic Proper has been divided into surface and deep-water compartments, which obviously makes the performance more realistic compared with the conventional whole-basin approach.

The budgets formed demonstrate that the estuarine sub-basins (Bothnian Bay, Gulf of Finland, Gulf of Riga) that have relatively short residence times are sources of nitrogen for the neighbouring basins, i.e. the annual amounts of sedimented and denitrified N are clearly smaller than riverine and atmospheric N loads into these basins (**Tables 3.1** and **3.2**). Similarly, the Danish Straits, which are a flow-through area between the Baltic Proper and the Kattegat, are not able to retain amounts comparable to their whole external nitrogen load. On the other hand, the Bothnian Sea is able to retain and remove amounts of N that are comparable to the total net loads into the sea area, including both external loads and inputs from the neighbouring basins. This seems to be valid also for the Baltic Proper, despite its

Table 3.1 **External (riverine and atmospheric) loads of total N (t yr^{-1}, upper figure) and estimated total sinks (sediment burial and denitrification, t yr^{-1}, lower figure in bold) in the various budget studies.**

Reference, calculation years/ sea area	Bothnian Bay, BB	Bothnian Sea, BS	Gulf of Finland, GF	Gulf of Riga, GR	Baltic Proper, BP	Danish Straits, DS	Kattegat, KT
Wulff & Stigebrandt (1989), 1977–1986	60,000 **−21,000**	85,000 **−127,000**			964,000 **−729,000**		
Perttilä et al. (1995), 1990			140,000 **−130,000**				
Wulff et al. (2001b), 1970–1991	62,000 **−23,000**	103,000 **−191,000**			871,000 **−1,229,000**		
Savchuk (2005) 1991–1999	55,000 **−15,000**	103,000 **−112,000**	153,000 **−86,000**	112,000 **−68,000**	765,000 **−751,000**	92,000 **−56,000**	88,000 **−77,000**
Savchuk & Wulff (2007), 1991–2002 (nutrient data), 1997–2003 (loads)	62,000 **−21,000**	89,000 **−99,000**	128,000 **−74,000**	90,000 **−48,000**	482,000 **−880,000**	74,000 **−43,000**	89,000 **−92,000**

Table 3.2 **The calculated advective net transports of total nitrogen (t yr^{-1}) between the Baltic Sea sub-basins. The arrows denote the positive flux direction between the basins, i.e. total P flux of −12,000 between the Bothnian Sea (BS) and the Baltic Proper (BP) indicates that the mean net annual transport of total N is 12,000 t from the Baltic Proper to the Bothnian Bay. Basin acronyms: see Table 3.1. SK= Skagerrak.**

Reference	BB ▸ BS	BS ▸ BP	GF ▸ BP	GR ▸ BP	BP ▸ DS	DS ▸ KT	KT ▸ SK
Wulff & Stigebrandt (1989)	31,000	−12,000	-	-	102,000	-	-
Perttilä et al. (1995)	-	-	68,000	-	-	-	-
Wulff et al. (2001b)	39,000	18,000	-	-	140,000	-	
Savchuk (2005)	40,000	31,000	67,000	44,000	156,000	191,000	202,000
Savchuk & Wulff (2007)	41,000	49,000	72,000	43,000	133,000	170,000	170,000

usually poor deep-water oxygen conditions that probably worsen conditions for effective coupled nitrification-denitrification at the sediment-water interface. There are, however, some gaps in our knowledge on the controls of nitrogen loss processes in the Baltic Sea (Vahtera et al. 2007).

According to the two available budget estimates for the whole Baltic Sea, an amount corresponding to 85–120% of the total land-based and atmospheric loads of nitrogen into the Baltic Sea is retained and removed before entering the Skagerrak (Savchuk 2005; Wulff et al. 2007; **Tables 3.1** and **3.2**). The calculations do not include N_2 fixation, the estimates of which are from about 100,000 t yr^{-1} to over 400,000 t yr^{-1} for the Baltic Proper alone (**Chapter 3.2.5**).

Regarding phosphorus budgets, deep-water oxygen conditions controlled by salinity stratification and the sedimentation of organic matter evidently are the critical factors influencing sediment retention efficiency in the different Baltic Sea sub-basins. Owing to relatively low autochthonous production, the lack of a permanent halocline and good near-bottom oxygen conditions favouring coupled iron-phosphorus cycling, conditions in the Gulf of Bothnia are good for the effective retention of P (**Table 3.3**). According to the different budget estimates, the amount of P permanently retained is from 1.4 to 4 times the land-based P load into the Bothnian Sea.

The Baltic Proper - owing to its strongly restricted renewal of deep-water oxygen leading to reduced conditions and low P binding efficiency of the surface sediments - retains less than half of its external load, and is a net source of P for all its neighbouring basins except the Gulf of Riga (**Tables 3.3** and **3.4**). In the Gulf of Finland, the conditions have varied greatly. In the late 1990s, when oxygen conditions strongly worsened and the sediment release of P increased (Pitkänen et al. 2001), the Gulf of Finland turned from a net P importer to a net exporter (**Tables 3.3** and **3.4**). As a whole, the Baltic Sea retains about 60% of its external phosphorus load, i.e. a smaller proportion than in the case of nitrogen. The main reason for the difference is the poor ability of the Baltic Proper to retain P.

Table 3.3 External (riverine and atmospheric) loads of total P (t yr^{-1}, upper figure) and estimated total sinks (sediment burial or in case of a positive figure, flux from the sediment, t yr^{-1}, lower figure in bold) in the various budget studies.

Reference/ sea area	Bothnian Bay, BB	Bothnian Sea, BS	Gulf of Finland, GF	Gulf of Riga, GR	Baltic Proper, BP	Danish Straits, DS	Kattegat, KT
Wulff & Stigebrandt (1989), 1977–1986	3,300 **−2,400**	3,900 **−5,500**			59,300 **−36,700**		
Perttilä et al. (1995), 1990			11,800 **−20,000**				
Wulff et al. (2001b), 1970–1991	3,600 **−5,700**	4,500 **−13,800**			37,200 **−16,700**		
Savchuk (2005), 1991–1999	3,400 **−3,800**	3,700 **−18,600**	5,000 **−6,300**	2,400 **−1,500**	18,600 **17,700**	2,500 **−5,000**	2,100 **−3,200**
Savchuk & Wulff (2007), 1991–2002 (nutrient data), 1997–2003 (loads)	3,200 **−4,100**	3,600 **−8,500**	7,300 **−4,100**	2,500 **−400**	22,100 **−8,300**	1,700 **−200**	1,900 **−800**

Table 3.4 The calculated advective net transports of total phosphorus (t yr^{-1}) between the Baltic Sea sub-basins. The arrows between the basin acronyms denote the positive flux direction between the basins. Basin acronyms: see **Table 3.1.** SK= Skagerrak.

Reference	BB▸BS	BS▸BP	GF▸BP	GR▸BP	BP▸DS	DS▸KT	KT▸SK
Wulff & Stigebrandt (1989)	200	−3,900	-	-	5,200	-	-
Perttilä et al. (1995)	-	-	−8,000	-	-	-	-
Wulff et al. (2001b)	−100	−5,800	-	-	11,000	-	-
Savchuk (2005)	−300	−15,200	−1,300	800	20,500	18,100	17,000
Savchuk & Wulff (2007)	−900	−5,800	3,200	2,100	13,300	14,800	15,900

According to the load estimates presented by HELCOM (2005b), the gross loads of total N and total P within the Baltic Sea catchment decreased by about 380,000 t yr^{-1} of N and about 30,000 t yr^{-1} of P from 1985 to 2000. During the same period, the combined riverine load and atmospheric N deposition into the Baltic Sea decreased by about 300,000 t yr^{-1} (Vahtera et al. 2007), while the decrease in external P load from the latter half of the 1980s to 2000 appeared to be about 10,000 t yr^{-1} (Conley et al. 2002b, HELCOM 2004). The direct atmospheric N deposition alone decreased by about 100,000 t yr^{-1}, which is explained by decreased N emissions of 2.3 million t yr^{-1} in the Baltic Sea countries during the same period (HELCOM 2005a).

According to Conley et al. (2002a) and Vahtera et al. (2007), internal processes in the Baltic Sea strongly govern year-to-year variations in the pools of both inorganic N and P over inter-annual changes in external loads. Long-term data analyses by Fleming-Lehtinen et al. (2008) and in **Chapter 2.2** show decreasing trends since the mid-1980s for wintertime inorganic N and also to some extent for inorganic P, except for in the Gulfs of Finland and Riga. As the decreases seem to follow the decreased trends in external loads, the trend analyses give support to an assumption that over longer temporal scales (tens of years), the decreased concentrations and pools are associated with decreased external loads. Although a year-to-year reduction in the external load is small compared with year-to-year changes in the basin-wide pool, the long-term cumulative change will gradually decrease the pool as well as the fluxes to the neighbouring basins. Results of ecosystem modelling studies suggest the same (Pitkänen et al. 2007b, Wulff et al. 2007).

The large-scale balance of P is strongly affected by its long residence time in the Baltic Sea (Savchuk 2005), which is largely caused by the poor sediment retention ability of P especially in the Baltic Proper (see **Chapter 3.2.4**), while the balance of N is affected by denitrification and N_2 fixation in addition to sediment burial. Studies from the estuarine sub-basins of the Gulfs of Finland and Riga with relatively short residence times suggest downward inorganic N trends in the 1990s as a result of decreased external N loads, while at the same time inorganic P concentrations increased in both sub-basins despite decreased external loads, most probably owing to increased sediment release (Pitkänen et al. 2001, Yurkovskis 2004, Vahtera et al. 2007).

4 HOW ARE THE ECOLOGICAL OBJECTIVES FOR EUTROPHICATION BEING MET?

The eutrophication issues have been recognized by all Baltic Sea countries for decades and many plans have been agreed and implemented. A strong focus has been placed on actions leading to reductions from point sources as well as diffuse sources. The underlying principle has been the following: because increased loads result in nutrient enrichment and subsequently eutrophication, the loads have to be reduced to the same levels as in the time before eutrophication took place.

The central driver is clearly the well-documented eutrophication signal and symptoms in combination with a political will to improve the situation. In contrast, the legal drivers implementing the political decisions are a complex patchwork. Often, it is unclear which legal or political driver is the most ambitious or strict and often the drivers overlap in term of scope and geography.

A key point is that the combination of drivers work in the same direction and set a course that ultimately strives at a reduction of loads and improved eutrophication status. Consequently, major load reductions have been achieved, first of all in regard to point sources, cf. **Chapter 3**. In some parts of the Baltic Sea, improvements have been documented, e.g. in the Kattegat and Danish Straits (Carstensen et al. 2006). However, nitrogen and phosphorus loads have not been reduced to levels where eutrophication signals in general are nearing the overall objective of a Baltic Sea unaffected by eutrophication.

4.1 What has been done?

The work done so far has mainly been driven by the Helsinki Convention, European water quality directives, a number of international conventions and directives indirectly focusing on water quality, as well as nationally motivated actions.

4.1.1 HELCOM

The Convention on the Protection of the Marine Environment of the Baltic Sea area (Helsinki Convention) was first signed in 1974. The initial convention primarily concerned the prevention and elimination of pollution by hazardous substances and its focus was mostly on the open sea marine environment.

Assessments of the status of the marine environment of the Baltic Sea conducted by HELCOM provide insight into how eutrophication has gradually evolved to be the primary environmental problem in the Baltic Sea. The first, baseline assessment of the effects of pollution on the marine environment in 1980 recognized eutrophication but considered it as only partially caused by man (HELCOM 1980). The changes were considered to be due partly to natural long-term hydrographic changes and partly to increased anthropogenic nutrient loads. The degree, causes and effects of eutrophication and its relation to the oxygen depletion in the deep basins were unresolved issues at the time (HELCOM 1980).

The status assessments have relied on coordinated monitoring of the Baltic Sea carried out jointly by all Contracting Parties. A coordinated Baltic Sea monitoring programme was started in 1979. Joint monitoring consisted of coordinated monitoring of physical, chemical, and biological variables and today it still serves the data needs of the assessments produced by HELCOM.

The first comprehensive assessment of the status of the marine environment covered the years 1980–1985. It concluded that trends in pollution by hazardous substances - the initial focus of HELCOM - were decreasing, while signs of eutrophication were clearer than previously (HELCOM 1987a). Anthropogenic sources were suggested to be partly responsible for the increasing nutrient concentrations in the surface layers although a relation to changes in salinity of the Baltic Sea was also considered responsible. The second assessment of the state of the marine environment of the Baltic Sea, covering data from 1984–1988, concluded that in the 1980s the nutrient concentrations were at a high level and allowed phytoplankton to flourish (HELCOM 1990). The Third Periodic assessment for 1989–1993 recorded continuation of the increase of nutrient concentrations in most parts of the Baltic Sea (HELCOM 1996). For the first time, this assessment recognized that nutrients in runoff from arable land may have been a significant source for eutrophication.

From a policy perspective, HELCOM responded to this new understanding of the anthropogenic sources of eutrophication of the sea by adopting various Recommendations in the latter half of the

1980s, aimed at limiting nutrient pollution from municipal wastewater treatment plants, agriculture and industry. In the 1990s, HELCOM Recommendations concerning reduction of air emissions from ships and nutrient discharges from fish farming and forestry were also adopted.

Nutrient pollution was for the first time strategically addressed at a higher political level by the Ministers of the Environment of the Baltic Sea States at a Ministerial Meeting held in 1988. The resulting 1988 Ministerial Declaration stated that *'…efforts on reduction of the load of pollutants should aim at a substantive reduction of the substances most harmful to the ecosystem of the Baltic Sea, especially of heavy metals and toxic and persistent organic substances, and nutrients for example in the order of 50 percent of the total discharges of each of them, as soon as possible, but not later than 1995'*. In a sense, the 1988 Ministerial Meeting was a turning point in the history of the Helsinki Commission and served as a beginning for more advanced environmental policy (Lääne et al. 2002). In 1992, the Helsinki Convention was revised to embrace new environmental principles and the changed geopolitical situation. The revised convention also became more explicit with regard to combating eutrophication, including now also the new Annex III specifically addressing the need for reduction of nutrient loads originating from municipal wastewaters and agriculture.

In order to support the implementation of the reduction targets agreed upon at the 1988 Ministerial Meeting by further reducing loads of organic matter, nutrients and other harmful substances, the Baltic Sea Joint Comprehensive Environmental Action Programme (JCP) was established in 1992. Identification and elimination of pollution Hot Spots was an important part of this work, and initially 132 Hot Spots were identified, of which some were municipal wastewater treatment plants and agricultural 'sites'.

The assessment of the status of the marine environment for 1994–1998 recorded positive signals, such as decreased phosphorus concentrations in some areas, mainly owing to improved wastewater treatment (HELCOM 2001). At the same time, however, eutrophication in the Gulf of Finland was aggravated by changes in hydrographic conditions in the Gulf.

HELCOM pollution load compilations assessing nutrient loads have evolved from the first attempt that was based on fragmented and largely incomparable data (HELCOM 1987b) to the fourth pollution load compilation (PLC-4) which employed a commonly agreed approach and guidelines to

BOX 7: Comparison of the 50% reduction target with the BSAP

In September 1988, the Ministers of Environment of the Baltic Sea States decided that nutrient discharges should be reduced by 50% from late 1980s levels by the year 1995 (The 1988 Ministerial Declaration). This target concerned reduction of nutrient discharges from sources, mainly point sources but also diffuse sources. Airborne pollution was also to be minimized. The Baltic Sea Action Plan (HELCOM BSAP), accepted in 2007, sets sub-basin and country-wise nutrient reduction targets to achieve good ecological status of the Baltic Sea.

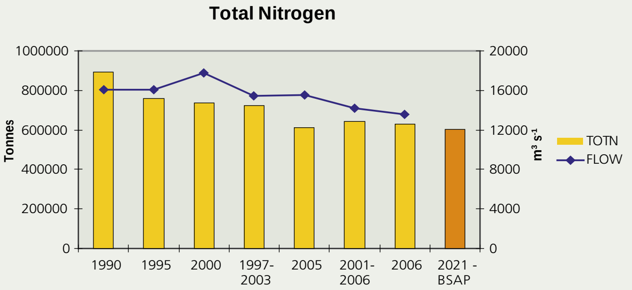
According to the report ‘Evaluation of the implementation of the 1988 Ministerial Declaration regarding nutrient load reductions in the Baltic Sea catchment area’ (Lääne et al. 2002) regarding point sources, the 50% reduction target was achieved for phosphorus by almost all Baltic Sea countries, while most countries did not reach the target for nitrogen. The results also showed that measures to reduce nutrients from agriculture failed widely.

Total nitrogen[1] and phosphorus loads entering the Baltic Sea (as riverine and direct point-source discharges) amounted to 891,000 t and 51,100 t, respectively, in 1990 according to the Second Baltic Sea Pollution Load Compilation (PLC) (HELCOM 1993), which in this review represented fairly well the levels that prevailed in the late 1980s, as referred to in the Ministerial Declaration. The maximum allowable nutrient input targets in the Baltic Sea Action Plan for the whole Baltic Sea are 41% of the 1990 load of phosphorus and approximately 68% of that of nitrogen.

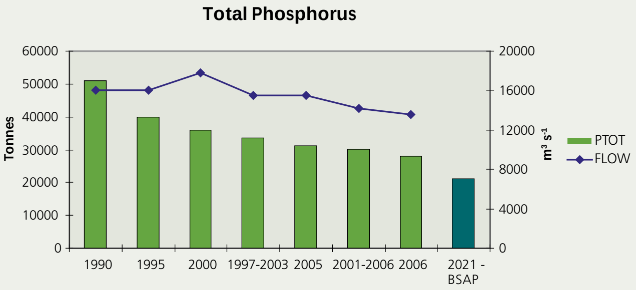
By the year 1995, the total waterborne phosphorus load had decreased by around 22% to 39,850 t according to the annual PLC reporting to HELCOM. The total nitrogen load decreased at the same time by 15% to around 761,000 t. These results support the above-mentioned decrease in P discharges to the catchment from point sources. The fact that the decrease was only 22% in nutrient discharges to the Baltic Sea is most probably a consequence of the poor results achieved in efforts to reduce leaching of phosphorus from agriculture, which comprises about half of the total anthropogenic P load into the catchment of the Baltic Sea. In direct discharges from point sources to the sea, the decrease was almost as evident as in the reported loads to the catchment. From 1990 to 1995, the total P load decreased by 41% and the total N load by 20%. During the decade from 1990 to 2000, the direct point source phosphorus load went down by 68% and the nitrogen load by 60%. Comparison of those figures with the results achieved in the catchment is more relevant because both represent loads at sources. These results indicate that the claim by most of the Contracting Parties that they have reached the initial 50% reduction target for point sources is reasonable.

If the original target in the 1988 Ministerial Declaration had been set for the total nutrient load entering the Baltic Sea instead of sources in the catchment, the 50% reduction target would not yet have been achieved. For phosphorus, the reduction was 45% by the year 2006, for nitrogen only 30%. In addition, this observed decrease is partly explained by the lower runoff in 2006.

The achievement of the contemporary BSAP target load for phosphorus (21,000 t) by the year 2021 will mean reaching the target set by the ministers in 1988 also in relation to the direct load into the Baltic Sea. Reaching that level, however, also requires a significant reduction in the diffuse load from agriculture. In the case of waterborne nitrogen, the original 50% reduction target will not be achieved but according to the model approach, that is not necessary in order to fulfil the target of good ecological status in the open parts of the Baltic Sea.



Panel A: Direct riverine and point-source loads of nitrogen to the Baltic Sea. The maximum allowable loads *sensu* the BSAP are indicated in orange.



Panel B: Direct riverine and point-source loads of phosphorus to the Baltic Sea. The maximum allowable loads *sensu* the BSAP are indicated in dark green.

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[1] Total nitrogen load has been corrected for Latvia, Lithuania and Russia. Originally only dissolved nitrogen in rivers was reported. Those figures have been multiplied by 1.6. Missing figures for unmonitored rivers and coastal areas estimated on the basis of data from the years 1995 and 2000.

quantify loads and estimate the source-apportionment of the loads (HELCOM 2004). PLC-4 clearly indicated that losses from diffuse sources, including agriculture and forestry, were the main sources of excessive loads of both nitrogen and phosphorus entering the Baltic Sea in 2000. PLC-4 further emphasized the great importance of having accurate estimates of diffuse loads of nutrients entering the Baltic Sea in order to be able to assess the effectiveness of policy measures towards preventing nutrient pollution.

In 2002, an evaluation of achievements revealed that the 50% reduction target for the time period from 1987 to 1995 had been achieved for phosphorus discharges from point sources by almost all countries, while most countries had not reached the targets for nitrogen (Lääne et al. 2002). Agricultural loading levels showed smaller decreases than point-source loading despite the fact that almost all countries in transition had achieved the 50% target. On the other hand, accurate estimations of changes in agricultural loading were hampered by a lack of direct monitoring data. Further estimation of achievements between 1985 and 2000 showed that as a result of improved treatment of industrial and municipal wastewaters, nutrient discharges from point sources had decreased significantly. However, the reduction targets for diffuse sources such as agriculture had not yet been fulfilled (Knuuttila 2007).

It was clear that eutrophication remained a major environmental problem in the early 2000s. The HELCOM Bremen Ministerial Meeting Declaration of 2003 demanded further actions, in particular in the agricultural sector, to reduce diffuse nutrient loads. In addition, HELCOM was tasked to implement an ecosystem approach to the management of human activities and the idea of developing ecological objectives with indicators was put forward. As a result, in 2006 HELCOM adopted a system of ecological objectives with the specific strategic goal for eutrophication ‘Baltic Sea unaffected by eutrophication’ defined by five specific ecological objectives (HELCOM 27/2006, document 2/6) (see the introductory part of **Chapter 2**).

To have a more targeted approach to addressing the symptoms of eutrophication, nutrient reduction targets taking ecosystem functioning as well as sub-regional differences into account were considered necessary. A model-based approach employing sub-regional targets related to selected ecosystem features such as water transparency was established (e.g. Wulff et al. 2007).

Following the principle of adaptive management and in order to implement the ecosystem approach to the management of human activities, HELCOM began elaborating the HELCOM Baltic Sea Action Plan (BSAP) which was eventually adopted in November 2007 (HELCOM 2007b). The starting point for the BSAP eutrophication work was the outcome of the HELCOM EUTRO project, which indirectly defined primarily quantitative targets for eutrophication-related ecological objectives reflecting good ecological/environmental status with regard to eutrophication (HELCOM 2006), and modelling efforts by the Swedish MARE program (e.g. Wulff et al. 2007). Initial estimates of the quantities of nutrient reductions needed to reach the target levels for eutrophication were produced by the MARE program. In addition, scenarios were elaborated on how far the full implementation of existing HELCOM Recommendations, as well as EU legislation and programmes, would bring the Baltic Sea towards the agreed ecological objectives for eutrophication, using the target ‘Clear water’ as a basis. These results produced by MARE were used to develop specific reduction targets and actions related to reducing nutrient loading to the Baltic Sea in the BSAP. The BSAP defines maximum allowable nutrient loads that will provide achievement of the eutrophication targets for the whole Baltic Sea and each of its sub-basins. The required reductions in nutrient loads were estimated based on the maximum allowable nutrient loads and average nutrient load levels from 1997 to 2003. Required reductions of annual loads addressed to the whole Baltic were estimated as 15,250 tonnes phosphorus and 135,000 tonnes nitrogen. Similarly, quantitative reduction requirements were addressed to each of the sub-basins and provisional allocations of nutrient reduction requirements to each HELCOM country and to transboundary loads were included in the Action Plan.

The idea behind the allocation system was that maximum allowable nutrient loads and country-wise reduction allocations allow the Contracting States more flexibility in choosing the management actions that are nationally most suitable and cost-effective to reach the agreed targets.

The Action Plan also includes direct measures to address nutrient pollution. The strengthening of Annex III of the Convention is concerned mainly with the prevention of nutrient releases from livestock production, including requirements for environmental permits for animal enterprises. For wastewater treatment, it was estimated that with stricter treatment it would be possible to further reduce phosphorus loads to the Baltic Sea by 2,000 tonnes per year. Consequently, a HELCOM Recommendation concerning municipal wastewater treatment and another Recommendation concerning on-site wastewater treatment for single-family homes, small businesses and settlements of less than 300 person-equivalents were adopted.

In addition to addressing agricultural nutrient loads and municipal wastewaters, the BSAP places further emphasis on measures to reduce atmospheric deposition of nitrogen. According to HELCOM scenarios, deposition of nitrogen to the sea would not decrease even if the existing targets for nitrogen in the United Nations Economic Commission for Europe (UNECE) Gothenburg Protocol (UNECE 1999) and the EU National Emission Ceiling (NEC) Directive (Anon. 2001) were reached. Therefore, the Action Plan requires Contracting Parties to work towards strengthened targets to be adopted within these organizations. To address airborne nitrogen emissions from shipping, with the Action Plan HELCOM countries committed themselves to evaluating Baltic Sea-specific environmental effects and the sufficiency of the proposed new NOx emission control measures of the MARPOL 73/78 Annex VI (Anon. 1978). Nutrient emissions from shipping are also addressed in the BSAP in the sense that the Contracting Parties will produce a joint Baltic proposal within the International Maritime Organization (IMO) to amend regulations under Annex IV of the same convention to eliminate discharges of sewage from ships in the Baltic, initially from passenger ships and ferries.

Although it was recognized that the maximum allowable nutrient loads and the country-wise allocations of the BSAP were based on the best knowledge at the time and that reviewing and revising of the figures should start as soon as the BSAP was adopted, the Action Plan for the first time addresses eutrophication with a holistic ecosystem approach.

4.1.2 European water quality directives

European policies on water quality are, *inter alia*, aimed at the mitigation of eutrophication. In this section, European directives in the context of nutrient enrichment and eutrophication are described. In the Baltic Sea area, these directives concern all countries with the exception of the Russian Federation.

Urban Waste Water Treatment Directive (UWWTD)

The objective of the Urban Waste Water Treatment Directive (Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment; Anon. 1991a) is to protect the environment from the adverse effects of discharges of wastewater. The directive concerns the collection, treatment and discharge of urban wastewater and the treatment of discharges of wastewater from certain industrial sectors. The degree of treatment (i.e. emission standards) of discharges is based on an assessment of the sensitivity of the receiving waters. Member States shall identify areas that are ‘sensitive’ in terms of eutrophication. Competent authorities shall monitor discharges and waters subject to discharges.

Each Member State can chose between two options of implementation. Either the Member State declares the whole country as a sensitive area and consequently has to implement the directive for its entire territory or sensitive areas have to be designated and the implementation is restricted to them. Those coastal states of the Baltic Sea which joined the EU in 2004 negotiated transition periods for the implementation of this directive which extend to 2015.

Nitrates Directive (ND)

The objective of the Nitrates Directive (Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agriculture; Anon. 1991b) is to reduce water pollution caused or induced by nitrates from agricultural sources and to prevent further such pollution. EU Member States shall designate ‘vulnerable zones’, which are areas of land draining into waters affected by pollution and which contribute to pollution. Member States shall set up, where necessary, action programmes promoting the application of the codes of good agricultural practices. Member States shall also monitor and assess the eutrophication status of freshwaters, estuaries and coastal waters every four years.

Each Member State can chose between two options of implementation. Either the Member State declares the whole country as a vulnerable zone and consequently has to implement the directive for its entire territory or vulnerable zones have to be designated and the implementation is restricted to them.

The Water Framework Directive (WFD)

In 2000, the European Parliament and the Council adopted the EU Water Framework Directive (WFD), which provides a framework for the protection of groundwater, inland surface waters, transitional waters (e.g. estuaries), and coastal waters. The overall aim of the WFD is: (1) to achieve a good ecological status and to prevent further deterioration, protect, and enhance the environmental status of aquatic systems, and (2) to promote the sustainable use of water, while progressively decreasing or eliminating discharges, losses and emissions of pollutants and other pressures for the long-term protection and enhancement of the aquatic environment.

The WFD is intended to improve the ecological status, including eutrophication status, of all European surface waters, many of which are considered to be eutrophic. The directive provides national and local authorities with a legislative basis for the maintenance and recovery of water quality to achieve good ecological and chemical status for all surface waters and good chemical status for groundwater. Accordingly, the directive can be considered the most significant piece of legislation of the past 20 years in regard to water policy, not only in Europe but also by non-European countries viewing EU legislation as a benchmark for their own legislation.

Marine Strategy Framework Directive (MSFD)

In 2005, the European Commission proposed an ambitious strategy and a directive to protect in a targeted manner the marine environment across Europe. The Framework for Community Action in the field of Marine Environmental Policy (Directive 2008/56/EC, Marine Strategy Framework Directive, Anon. 2008) requires the Member States to take the necessary measures to achieve or maintain good environmental status in the marine environment by the year 2020. The Directive will constitute the environmental pillar of the future European maritime policy designed to achieve the full economic potential of oceans and seas in harmony with the marine environment.

The Directive establishes Marine Regions as implementation units. Each Member State, in cooperation and coordinating with other Member States and third countries within a Marine Region, will be required to develop Marine Strategies for their marine waters. For achieving the coordination with other countries in the region, the Member States are to use existing regional cooperation structures, including Regional Seas Conventions. The Marine Strategies will contain a detailed assessment of the state of the environment, including an analysis of pressures and impacts on the marine environment, a determination of ‘good environmental status’ at regional level, and the establishment of environmental targets with associated indicators and monitoring programmes. Eutrophication is specifically addressed in one of the quality descriptors of Annex I of the Directive. Each Member State will draw up and operationalize a programme of measures. Impact assessments, including detailed cost-benefit analyses of the measures proposed, will be required prior to the introduction of any new measure. Where it would be impossible for a

Member State to achieve the level of ambition of the environmental targets set, special areas and situations will be identified in order to devise specific measures tailored to their particular contexts. In the marine regions where the status of the sea is so critical that it necessitates urgent action, the Member States should devise a plan of action with programmes of measures that have an earlier entry into operation and the Commission should consider providing supportive action for the enhanced efforts of the Member States by designating the region as a pilot project.

The Directive is consistent with the Water Framework Directive, which requires that surface freshwater and groundwater bodies (lakes, streams, rivers, estuaries, coastal waters) achieve a good ecological status by 2015 and that the first review of the River Basin Management Plans should take place in 2021.

Inter-relations of the directives

The Baltic Sea Action Plan and the directives are closely linked, especially the Action Plan and the recently adopted Marine Strategy Framework Directive. The ecological objectives *sensu* the BSAP are tightly connected to the overarching goal of 'good environmental status' *sensu* the MSFD as well as the goal of 'good ecological status' *sensu* the WFD. Furthermore, in regard to eutrophication the WFD is tightly connected to the UWWTD (sensitive vs. non-sensitive waters) and ND (polluted vs. unpolluted waters). The relationships in terms of management standards and human pressures are outlined in **Fig. 4.1**.

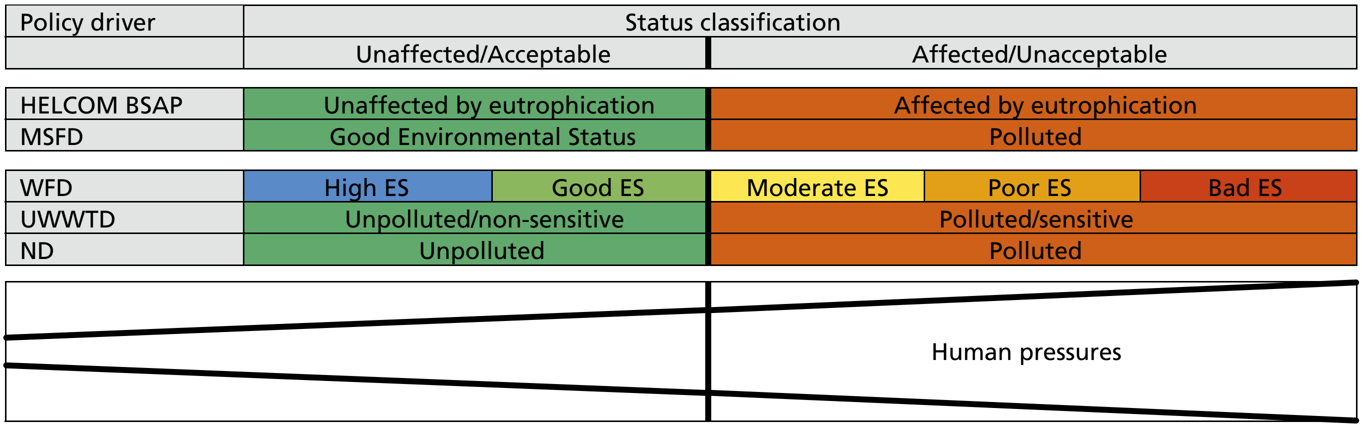
4.1.3 International conventions and other relevant European directives

Improvement of water quality in Europe is indirectly supported by the implementation of international conventions and a suite of European directives focusing on emissions to air, integrated pollution prevention and control, and nature protection.

UN Conventions

The UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP), signed in Geneva in 1979, aims to reduce emissions contributing to transboundary air pollution in the UNECE region through coordinated efforts on research, monitoring and the development of emission reduction strategies on regional air pollution and its effects. The Convention has been extended by eight protocols that identify specific measures to be taken by the 51 Member Parties to cut their emissions of air pollutants. An important factor in the development of the Convention has been the elaboration of the effects-based approach as well as the increased use of integrated assessment models as a basis for policy-making. The aim of this approach is to promote an international cost-effective and effect-oriented policy.

The International Convention for the Prevention of Pollution from Ships, 1973, as modified by the Protocol of 1978 relating thereto (MARPOL 73/78), is the main international convention covering prevention of pollution of the marine environment by ships. The Convention includes regulations aimed at preventing and minimizing



Policy driver	Status classification				
	Unaffected/Acceptable		Affected/Unacceptable		
HELCOM BSAP	Unaffected by eutrophication		Affected by eutrophication		
MSFD	Good Environmental Status		Polluted		
WFD	High ES	Good ES	Moderate ES	Poor ES	Bad ES
UWWTD	Unpolluted/non-sensitive		Polluted/sensitive		
ND	Unpolluted		Polluted		

Figure 4.1 Relationships between the Baltic Sea Action Plan and some key European water policy directives with direct focus on eutrophication status. BSAP = Baltic Sea Action Plan; MSFD = Marine Strategy Framework Directive; WFD = Water Framework Directive; UWWTD = Urban Waste Water Treatment Directive; ND = Nitrates Directive; ES = Ecological Status *sensu* the WFD. Based on HELCOM (2006).

pollution from ships—both accidental pollution and that from routine operations—and currently includes six technical annexes. MARPOL 73/78 Annex IV deals with the prevention of pollution from ships by sewage. The Baltic Sea countries will have a joint submission to IMO in 2009 to amend this annex to establish the Baltic Sea as a special area where specific restrictions on the discharge of nutrients in sewage would need to be applied by passenger ships. MARPOL 73/78 Annex VI deals with preventing air pollution by ships. The regulations in this annex set limits on sulphur content in ships' fuel and nitrogen oxide emissions from ship exhausts and prohibit deliberate emissions of ozone-depleting substances. Annex VI has undergone a comprehensive review by the IMO Marine Environment Protection Committee (MEPC) to take into account current technology and the need to further reduce air pollution from ships. This review was completed in 2008. All HELCOM countries except one have ratified MARPOL 73/78 and the Annexes mentioned here. According to the Baltic Sea Action Plan, the Russian Federation is committed to ratify Annex VI by 1 January 2010.

National Emission Ceilings Directive (NECD)

The National Emission Ceilings Directive (Directive 2001/81/EC; Anon. 2001) sets pollutant-specific emission ceilings for each Member State to be met by 2010. It covers the pollutants SO_2, NOx, volatile organic compounds (VOCs) and NH_3 responsible for acidification, eutrophication, and the formation of ground-level ozone. The NEC Directive required Member States to draw up national programmes by 2006 in order to demonstrate how they were going to meet the national emission ceilings (NECs) by 2010. Member States are obliged each year to report their national emission inventories and projections for 2010 to the European Commission and the European Environment Agency.

Especially for NOx, the NEC targets are very ambitious, so that additional reduction measures are required in many Member States. NOx emissions could be further reduced by technical and non-technical measures in the transport sector, with the largest reduction potentials in road freight transport. Minor reduction potentials could be realized by ambitious control technologies for stationary sources.

Directive on Integrated Pollution Prevention and Control (IPPC)

With the adoption in 1996 of Directive 96/61/EC on Integrated Pollution Prevention and Control (IPPC, Anon. 1996), the European Union has taken an important step in pollution abatement. The IPPC Directive requires the reduction at source of discharges and emissions of pollutants (including hazardous substances) through the application of best available techniques (BAT).

After a two-year review process, the Commission adopted on 21 December 2007 a proposal for a new directive on industrial emissions in the form of a 'recast'. The new directive will integrate the IPPC Directive and six sectoral directives (concerning large combustion plants, waste incineration and co-incineration plants, certain solvent-using installations and TiO_2-producing installations). The new directive aims to achieve a complete and better-harmonized implementation of BAT. An important contribution to reduce pollutants responsible for eutrophication could result, for example, from the following proposals: (1) Article 16 (4) of the Commission's recast proposal requires the application of BAT to the spreading of livestock manure and slurry of IPPC installations, (2) Annex I of the proposal, defining the activities covered by the IPPC approach, requires in number 6.6 (intensive rearing of poultry or pigs) also that installations with fewer places for certain poultry species shall be covered by the new directive, and (3) furthermore, Annex I of the proposal also shall include the off-site treatment of wastewater not covered by Council Directive 91/271/EEC of 21 May 1991 concerning urban wastewater treatment and discharged by an installation covered by the new directive.

On the other hand, the Commission's proposal also contains significant disadvantages, for example: (1) the add-up-rule for installations rearing different types of species covered by the directive is insufficient; it is based on vague and difficult-to-assess nitrogen excretion factors instead of the fraction of animal places, and (2) the current proposal does not address the aquaculture sector.

The Habitats Directive (HD)

The Habitats Directive (Anon. 1992) is indirectly linked to nutrient enrichment and eutrophication, because the objective is to contribute towards

ensuring biodiversity through the conservation of natural habitats and of wild flora and fauna in the European territory of the Member States. Measures shall be designed to maintain or restore, at favourable conservation status, natural habitats and species of wild flora and fauna of interest. The habitats and species protected are identified and defined in Annexes I and II, respectively. Many coastal waters in Northern Europe are identified as eutrophic owing to anthropogenic loads. Member States are required to implement management plans in order to restore these coastal waters and to achieve a favourable conservation status. Member States shall monitor habitats and species with particular regard to priority habitat types and priority species.

Regulation (EC) No 648/2004 on detergents concerning the use of phosphates

At present, the EC is evaluating whether restrictions on phosphates in detergents are justified at EU level. The decision will be taken once sufficient evidence has been acquired and various policy options have been assessed in consultation with the Detergents Working Group. An impact assessment was initiated in 2007 with the aim of concluding it in 2008. The Commission will present a legislative proposal without delay once a decision is taken that restrictions are justified.

4.1.4 National actions to reduce nutrient loads

The countries located within the catchment area of the Baltic Sea have worked for decades on a political and practical level to reduce discharges, emissions, and losses of nutrients with the ultimate aim of improving the eutrophication status in the Baltic Sea. The main drivers have been the Helsinki Convention and its HELCOM Recommendations, as well as, for those states belonging to the European Union, European directives directly or indirectly focusing on nutrient enrichment and water quality. Both European directives and HELCOM Recommendations are put into operation by national laws and often implemented via national action plans. This section describes the national actions taken in each country with regard to the reduction of nutrient loads to the Baltic Sea.

Denmark

For more than 25 years, a great deal of work has been done to reduce coastal eutrophication in Danish marine waters. Emphasis has been placed on reductions of nutrient loads to coastal and open waters through a series of national action plans, and implementation of international marine conventions and EU legislation (Danish EPA 2000; Ærtebjerg et al. 2003).

The first national initiative adopted by the Danish Parliament to improve the aquatic environment was the 1984 NPo Action Plan. This plan implemented a suite of measures relating to the discharge of nitrogen (N), phosphorus (P) and organic matter (o) in wastewater and from agriculture. The means included discharge standards, a ban on direct discharges from farms, a ban on the application of manure on frozen or sloping fields, and regulation of animal density.

In 1986, widespread hypoxia occurred in the Danish Straits. The Danish Parliament adopted an agenda urging the Government to reduce the discharges and losses of nitrogen by 50% and phosphorus by 80% from agriculture, urban wastewater plants, and separate industrial discharges. This strategic aim was transformed into sector-specific reduction objectives, reduction percentages and discharge targets. These were established for the three most important sources: (1) discharges and losses from agriculture, (2) discharges from municipal wastewater treatment plants, and (3) discharges from industries with separate discharges. Annual losses of nitrogen from cultivated fields and direct discharges from farms were, according to Action Plan I, to be reduced from approximately 260,000 t to 133,000 t. The reduction objective was 127,000 t, of which ca. 100,000 t were from the fields and ca. 27,000 t from the farms. Annual discharges of phosphorus from agriculture were to be reduced from ca. 4,400 t to 400 t. Losses of phosphorus from fields were not included owing to the large uncertainty associated with the estimation of the loss. The annual discharge of nitrogen from municipal wastewater treatment plants was to be reduced from ca. 18,000 t to 6,600 t, while annual discharges of phosphorus were to be reduced from ca. 4,470 t to 1,220 t. Separate discharges of nitrogen and phosphorus from industries were to be reduced from ca. 5,000 t to 2,000 t for nitrogen and from 1,250 t to 200 t for phosphorus.

The reduction targets for both municipal wastewater treatment plants and industries were met in 1995, but the specific objectives and targets for agriculture were difficult to meet within the original time limit. A number of amendments to Action Plan I have been adopted by the Parliament. In 1991, the Action Plan on Sustainable Agriculture was adopted, focusing on the reduction of losses from fields. New measures were adopted in order to meet the objectives of Action Plan I. The Action Plan on the Aquatic Environment II was adopted in 1998 following a major hypoxia in Mariager Fjord in 1997 and the requirements of the Nitrates Directive. Action Plan II focuses reduction on nitrogen losses from fields, cf. the objective of Action Plan I. Action Plan III was adopted in 2004 and focuses on losses of phosphorus from agriculture, but not on further reductions in nitrogen losses.

The Danish achievements, which have led to reduced nutrient concentrations in coastal waters, are currently the first-ever nation-wide effort that has succeeded in managing and reducing loads from both point and diffuse sources (Carstensen et al. 2006; Kronvang et al. 2008).

The Danish Government considers eutrophication to be a serious environmental problem and, consequently, has set an agenda that will ultimately lead to an evaluation of Action Plan III and subsequently an Action Plan IV. This announced fourth national Action Plan on the aquatic environment is anticipated to amalgamate implementation of the WFD, the MSFD and the Baltic Sea Action Plan with a so-called Green Growth Strategy.

Estonia

Protection and restoration of Estonian surface waters, including the waters of the Baltic Sea, are regulated by the Estonian Environmental Strategy and its implementation document, the Estonian Environmental Action Plan. The recent Action Plan covers the period 2007–2013 and is harmonized with relevant European environmental directives. The requirements of the EU WFD, UWWTD, Nitrates Directive and several international conventions, e.g. the Convention on Long-Range Transboundary Air Pollution (CLRTAP), MARPOL, etc., are specifically targeted in the Action Plan. There is no separate environmental strategy or policy document on combating eutrophication in the Baltic Sea, but in the framework of implementation of the HELCOM Baltic Sea Action Plan, an implementation committee has been established by the Ministry of Environment with activities also listed in priority areas of combating eutrophication in the Baltic Sea. Reductions in the amounts of nutrients supplied to the Baltic Sea ecosystem are planned in close cross-sectoral cooperation and integration of activities and by the introduction of limitations and regulations on sources in agriculture, and municipal and industrial wastewater treatment sectors.

Estonia is finalizing its first action plan for implementation of the HELCOM BSAP in 2008–2011, under which several measures have been planned via ministerial cooperation to establish a legislative framework to implement actions to combat eutrophication in the Baltic Sea. These actions are related to strengthened EU directive requirements for municipal wastewater plant regulations as well as single-house wastewater management regulations, so-called point sources; diffuse source nutrient loads are also planned to be regulated through the revised Annex III of the Helsinki Convention. The Estonian BSAP Implementation Program 2008–2011 is planned to be put into effect at the end of autumn 2008.

Finland

To enhance the protection of the Baltic Sea and inland waters, Finland has developed a national Action Plan, which was adopted in 2005 and is currently in the implementation phase (Ministry of the Environment 2005). The Action Plan is based on Finland's programme for the protection of the Baltic Sea in 2002 (Ministry of the Environment 2002). The Plan is to be implemented by 2015, but many measures are continuous in nature. Concrete actions have also been elaborated in the Finnish Government decision-in-principle on Water Protection Policy Outlines to 2015. Other relevant national programmes for the protection of the Baltic Sea environment include the National Strategy and Action Plan for the Conservation and Sustainable Use of Biodiversity in Finland 2006–2016 and the Finnish Coastal Zone Strategy.

The national Action Plan has been jointly prepared by various administrative sectors and other actors. The plan is organized to encompass five main themes: 1) combating eutrophication, 2) reducing risks caused by hazardous substances, 3) reducing the harmful impacts of the use of the Baltic Sea, 4) preserving and increasing biodiversity, and 5) increasing environmental awareness. All sections are coordinated with HELCOM activities to ensure the national Action Plan's compatibility and applicability for joint efforts under the auspices of HELCOM and the EU in protection of the Baltic Sea marine environment in the coming years.

Combating eutrophication includes complex and multifaceted measures to reduce nutrient loads entering the Baltic Sea that cause undesirable eutrophication effects. The measures are targeted at reducing nutrient loads from agriculture, municipal wastewater, rural settlements, and industry. Nutrient loads from shipping, atmospheric loads and loads from neighbouring countries are also included. Target nutrient loads for various sectors have been defined in the Action Plan, with a requirement to comply with targets by 2015–2021. These target loads will fulfill the national nutrient reduction requirements implied by the BSAP.

In Finland, the implementation of the BSAP is carried out by continuous implementation of the above-mentioned national protection programmes and strategies and by implementing EU legislation on the national level. These actions cover nearly all the measures included in the BSAP. In particular, the measures implied by the BSAP have been integrated in WFD River Basin management plans. However, all national measures will be thoroughly evaluated and updated when needed to fully comply with the BSAP.

Germany

In the mid-1970s, about half of the phosphate load to German waters originated from washing and cleansing agents. In 1975, the consumption of phosphate due to its use in detergents amounted to 276,000 t yr^{-1} in Germany. This household consumption of phosphate decreased to currently 26,000 t yr^{-1} as a result of the Ordinance on Maximum Amounts of Phosphates in Washing and Cleansing Agents, effective since 1981, and the self-commitment of the German Cosmetic, Toiletry, Perfumery and Detergent Association (Industrieverband Körperpflege- und Waschmittel e.V., IKW) as of 1985. The reduction in phosphates from washing and cleansing agents as well as the improved treatment of sewage water led to a more than halving the overall phosphate loads to German waters.

In order to further reduce phosphate loads into waters, the Federal Ministry for the Environment, Nature Conservation and Nuclear Safety initiated in April 2007 a discussion with business and association representatives about possible reductions or substitutions of phosphate in dishwasher detergents and in industrial textile washing agents. On the European level, Germany champions the efficient limitation of phosphate in washing and cleansing agents within the further development of the EU Ordinance on Detergent 648/2004.

In the course of the implementation of the EC Waste Water Treatment Directive, Germany reached an important reduction of nutrient loads. Up to 81% of nitrogen and 90% of phosphorus loads have been reduced to date. Thus, the requirements of the ordinance regarding the elimination of nutrients have been met all over Germany, including the German part of the Baltic Sea catchment area.

Because phosphate-free textile detergents have been used in German households for many years, the potential to reduce nutrient loads from washing and cleansing agents is substantially smaller than for nutrients resulting from agriculture.

With regard to the implementation of the EC Nitrates Directive, the German government assumes that the area-wide implementation of the nitrates action programme (Ordinance on Fertilizers and regulations on manure tanks of the Federal States) is successful because nutrient loads have been demonstrably reduced. Further improvements regarding agricultural cultivation may be expected.

However, loads from agriculture may be reduced considerably despite the important progress in the implementation of the Nitrates Directive as well as the implementation steps of the Water Framework Directive (WFD) that have already been taken. Therefore, the German government considers the application of the nitrates action programme, amended in 2006/2007, to be necessary. Additionally, the refined measures for water pollution control specified in the programmes for agricultural development within the period between 2007 and 2013 must be put into practice. Furthermore, they have to be adapted to the findings of the evaluation and monitoring programmes of the WFD. Regarding the WFD, programmes of measures for the different river basins are in preparation. The German government believes that the above-mentioned activities together will bring forward water pollution control as a whole. They are also intended to guarantee the implementation of the measures and objectives described in the HELCOM Baltic Sea Action Plan (BSAP). However, the uncertainty resulting from much higher producer prices and their possible impact on the intensification of agricultural production and the cultivation of biofuel crops should be considered as well. Thus, it is quite important to observe area-wide high environmental standards for agricultural production in general and especially for the cultivation of biofuel crops.

The NEC Directive (2001/81/EC) requires EU Member States to draw up National Programmes, which include information on measures taken at national level to achieve the 2010 emission ceilings and to inform the public and the European Commission about these programmes. Germany drew up a National Programme in 2002, which was updated in 2006. The NEC Directive limits Germany's total national emission loads, *inter alia*, to 1,051 kt for nitrogen oxides (NOx) and 550 kt for ammonia (NH3). As stated in the National Programme 2006, Germany will not meet the ceilings for NOx and NH3 by 2010 without additional measures.

Latvia

Water protection, including marine waters, is the major priority of Latvian environmental policy. Priorities of the water sector are defined in the following policy planning documents: Latvian Sustainable Development Strategy, 2002; National Environmental Policy Plan (NEPP) for 2004–2008; National Programme on Biological Diversity, 1999; and the Environmental Monitoring Programme,

2006. Among the priorities set by the draft long-term Sustainable Development Strategy 'Latvia 2030' is the integration of environmental policy into sectoral policies, as well as the conservation of biodiversity to maintain the ecosystem of the Baltic Sea and its coastal zone and to improve their adaptation capacity to climate change. The NEPP for 2004–2008 (currently in force) defines the following policy objectives: to protect water ecosystems; to ensure the protection of marine waters, paying particular attention to the reduction of chemical contamination in the Baltic Sea and on the fulfilment of the international commitments of Latvia.

Implementation of EU directives will contribute to further reductions in nutrient loading to the Baltic Sea and combating eutrophication of the Baltic Sea. The 'Action Programme for vulnerable zones subject to special requirements for protection of waters and soil against pollution caused by nitrates from agricultural sources' was adopted in 2004. Only the central part of Latvia with the most intensive agriculture was designated as a vulnerable zone, assuming that in future this part of the country may be most relevant to the provisions of the Nitrates Directive.

Most of the activities prior to this assessment targeted the municipal wastewater sector. The 'Implementation plan for Council Directive 91/271/EEC Concerning Urban Waste Water Treatment' (UWWTD) was adopted in 2001. In accordance with the pre-accession agreement between Latvia and the EU, Latvia assumed a number of obligations towards the implementation of the UWWTD. By the end of 2008, Latvia is to complete improvements in the wastewater collection and drinking water supply systems in the largest cities with a population equivalent (p.e.) above 100,000 and, by the end of 2015 (according to exclusively agreed timetables), the requirements of UWWTD will be introduced in all agglomerations with a p.e. above 2,000. The Operational Programme 2007–2013, co-financed from the Cohesion Fund (CF) and the European Regional Development Fund (ERDF), provides support in a framework of seven priorities, *inter alia*, development of the infrastructure of water management, conservation of biological diversity, decreasing environmental risks, and development of the system of monitoring and control. By carrying out the listed activities, implementation of the requirements of the EU Directives and corresponding national legislation will be promoted.

Latvia achieved the overall HELCOM target of a 50% reduction of nutrient inputs already in 2003 and there are further plans to decrease pollution by 2010 (HELCOM 2003b). It should be noted, however, that more than half of the water entering the Baltic Sea and the Gulf of Riga originates beyond Latvian borders. Compared to the load at the river mouth, the transboundary pollution loads for nitrogen and phosphorus are, respectively, 63% and 60% in the Daugava, without taking into account riverine retention (HELCOM 2005b). Almost half of the pollution load in the Lielupe and the Venta originates outside Latvia. It could diminish Latvia's efforts to a large extent if further steps regarding transboundary pollution, involving HELCOM and the EU, are not considered.

The general future vision is to use the Baltic Sea Action Plan (BSAP) as a tool for the implementation of the EU Marine Strategy Framework Directive (MSFD) in Latvia. This has already started with the adoption of 'Guidelines for National Environmental Policy, 2009–2015' (Policy Guidelines) together with the Set of Environmental Indicators in 2008, taking into account requirements of the MSFD, WFD and BSAP. The Policy Guidelines outline the status, major pressures, and risks and set objectives and necessary actions in the field of marine environment protection according to the MSFD and BSAP. Eutrophication and cross-border pollution, including hazardous substances, are the central problem areas for action. An overall policy objective is to achieve good environmental status, including for marine biodiversity, in the marine waters under the jurisdiction of Latvia in the Baltic Sea and the Gulf of Riga by 2020.

Lithuania

Aiming towards good water quality in Lithuanian marine waters, as well as the Nemunas River, the Government of the Republic of Lithuania approved the 'Water quality improvement programme of the Curonian Lagoon' in 2006 for the 2006–2015 period. The programme was prepared jointly by various administrative sectors, including also the requirements of the WFD, Nitrates Directive, UWWTD and relevant national plans. The overall aim of this programme is to achieve good water quality in the Curonian Lagoon by 2015. The Curonian Lagoon, which is situated in the southeastern part of the Baltic Sea and belongs to Nemunas river basin, was taken as the subject of the Programme as it reflects the water quality of the Nemunas River and determines the water quality of Lithuanian marine waters, especially in the coastal zone. The programme was designed following integrated river basin management principles. Five objectives were determined: 1) to decrease water pollution from point sources; 2) to reduce pollution from diffuse sources; 3) to collect information and make assessments about surface water pollution in Nemunas basin caused by anthropogenic activities; 4) to determine environmental objectives for surface waters (including the Curonian Lagoon) within the Nemunas river basin and to set the measures to reach these objectives; and 5) to improve transnational cooperation in order to decrease pollution of the Nemunas River and the Curonian Lagoon. Measures, responsible institutions, the time period for implementation of measures, and expected results are described in the programme for each objective. Lower concentrations of nutrients and total biomass of phytoplankton in the lagoon, better conditions for the development of macrophytes, benthic invertebrates and ichthyofauna, and better water quality in the Baltic coastal zone will require a defined list of environmental objectives with programmes of measures and better cooperation with neighbouring countries for the reduction of pollution of the Nemunas and coastal waters. Such national action plans correspond to the tasks of EU directives and the BSAP.

Poland

In Poland, measures directed at the reduction of nutrient loads into the surface waters, inland and coastal marine waters are focused on two major areas: (a) implementation of the Nitrates Directive, and (b) a national programme of municipal wastewater treatment. Regarding the Nitrates Directive, the Directive 91/676 EC was transposed into national legislation in a number of detailed regulations. In all, 21 areas sensitive to nitrate of agricultural origin have been determined. The areas comprise ca. 2% of the total land area of Poland. For all areas, a programme of measures was established which includes the education of farmers regarding good agricultural practice, fertilization planning, technical aid and advice in manure storage construction and treatment of household wastewater, monitoring of the implemented practices, and surface and groundwater monitoring to control the effects of implemented measures. Regarding the national programme of municipal wastewater treatment, large investment efforts have been undertaken since 2005. The programme includes construction of a sanitary sewage system of 37,000 km length in 1557 settlements, comprising 76 agglomerations >100,000 p.e., 383 cities between 15,000–100,000 p.e. and 1,118 settlements between 2,000–15,000 p.e. The total cost of municipal wastewater treatment improvements in Poland between 2005 and 2015 will amount to 1.22×10^9 EUR. Both programmes are being implemented with substantial financial support from the EU.

Russia

The Russian Federation considers eutrophication as the most important problem in the Baltic Sea. The major sources of nutrient pollution of the Baltic Sea in Russia are the discharges of municipal wastewater from St. Petersburg, Kaliningrad and the Kaliningrad region. GUP Vodokanal of St. Petersburg has developed and endorsed a programme aimed at the reduction of discharges of untreated wastewater and implementation of a comprehensive phosphorus removal technology. The programme covers all issues related to wastewater discharge and treatment, including storm water. The programme includes the connection of direct outlets to the city wastewater ducts and treatment at WWTPs; the reconstruction of existing sewerage pipelines and wastewater treatment facilities; and an increase of capacity and improvement of the treatment process including stage-by-stage comprehensive removal of phosphorus to 1.0 mg l^{-1} and later to 0.5 mg l^{-1}. Considering the fact that in the coming 20 years the population of the southwest suburbs of St. Petersburg will almost double (the Kolpinsky, Pushkinsky, Pavlovsky and Petrodvortsovy districts' populations will increase from the current 308,000 people to 589,000 people in 2025), this programme also includes the construction of two new WWTPs that will operate as central WWTPs for those areas (Metallostroy and Lomonosov).

Development and implementation of a national Baltic Sea Action Plan (BSAP) is an integrated problem which will be solved at three levels: federal, regional and local. Currently, there are regional programmes (for the Kaliningrad region, Leningrad region, and Republic of Karelia) under development in which attention is given to building, reconstruction, and/or modernization of water supply and wastewater treatment systems and biodiversity preservation. These plans are being developed for the period up to 2015 and will be financed from federal and regional budgets. At the federal level by order of the Minister of Natural Resources and Ecology, an interdepartmental working group concerning implementation of the BSAP at federal level was approved in 2008. On termination of the issues connected with administrative reforms in the Government of the Russian Federation in May 2008, this group will continue the work. Great value is given to the realization of international bilateral and the multilateral projects whose basic components are the actions directed towards achievement of HELCOM BSAP purposes.

Sweden

The basis for the Swedish work on combating eutrophication in the Baltic is the Swedish Environmental Quality Objective 'Zero Eutrophication'. The objective envisages an environmental state that is relatively unaffected by eutrophication to be achieved by the year 2020. The objective defines the state of environment that the environmental policy aims to achieve and provides a coherent framework for environmental programmes and initiatives at national, regional and local level. Interim targets on cuts in emissions of ammonia, nitrogen oxides, phosphorus and nitrogen should be achieved by the year 2010. The interim target for nitrogen inputs to water is based on the HELCOM target of a 50% reduction. New interim targets for nitrogen and phosphorus inputs to the Baltic Sea have been proposed by the Swedish Environmental Objectives Council to the Swedish Government (to be achieved by the year 2016). The proposals are based on the Swedish reduction assignments that Sweden was given in the agreement of the HELCOM Baltic Sea Action Plan. Central environmental work - such as the Swedish Board of Agriculture's action programme, which has been in place since 1988 and where the current programme dates from 2000 - is being carried out to reach the targets. Actions are also being taken in the other sectors. Action programmes addressing eutrophication are also being prepared by the Swedish water authorities as a part of the implementation of the Water Framework Directive.

The Swedish government has given an assignment to the Swedish Environmental Protection Agency in collaboration with the Swedish Board of Agriculture and other concerned authorities to propose a national plan for Sweden. The first part of the assignment was reported in May 2008 and elaborates as far as possible concrete measures concerning eutrophication. Measures will be further elaborated in the second part of the assignment which should be reported in July 2009.

4.2 Which supplementary measures are needed?

The term 'supplementary measures' can be interpreted in somewhat different ways: (1) 'supplementary' in regard to those measures already taken, (2) 'supplementary' in the sense of 'planned', and (3) 'supplementary' in contrast to basic measures, e.g. reductions of emissions, discharges and losses. Basic measures for the reduction of eutrophication include those measures which have been implemented or planned, e.g. those in the HELCOM Baltic Sea Action Plan and

Table 4.1 Maximum allowable annual loads of phosphorus and nitrogen to achieve 'good environmental status' (calculated for water transparency) and corresponding minimum load reductions (in tonnes) calculated per sub-basin (based on HELCOM 2007b).

	Maximum allowable nutrient loads (tonnes)		Inputs in 1997–2003 (normalized)		Needed reductions (interim allocation)	
	Phosphorus	Nitrogen	Phosphorus	Nitrogen	Phosphorus	Nitrogen
Bothnian Bay	2,580	51,440	2,580	51,440	0	0
Bothnian Sea	2,460	56,790	2,460	56,790	0	0
Gulf of Finland	4,860	106,680	6,860	112,680	2,000	6,000
Baltic Proper	6,750	233,250	19,250	327,260	12,500	94,000
Gulf of Riga	1,430	78,400	2,180	78,400	750	0
Danish Straits	1,410	30,890	1,410	45,890	0	15,000
Kattegat	1,570	44,260	1,570	64,260	0	20,000
Sum	21,060	601,710	36,310	736,720	15,250	135,000

those contained in the WFD and MSFD. Strictly speaking, supplementary measures are measures that can be envisaged in addition to those already agreed upon within the Baltic Sea Action Plan. The need for additional measures can be a result of incorrectly estimated basic measures or altered background conditions, requiring additional effort. Supplementary measures also include new innovations and technical solutions.

The HELCOM Baltic Sea Action Plan (BSAP) contains measures that in 2007 were estimated to be sufficient to reduce eutrophication to a target level that would correspond to good ecological and environmental status by the year 2021 (HELCOM 2007a). It was estimated that nutrient load reductions of 135,000 t of nitrogen and 15,250 t of phosphorus from average annual nutrient loads (based on loads during the period 1997–2003) would be needed. The main bulk of reductions were addressed to the Baltic Proper, while the Gulf of Bothnia was considered to be in good ecological/environmental status and thus not in need of reductions. It was estimated that the reductions would result in achieving the eutrophication-related targets on water transparency, primary production and nutrient concentrations (Wulff et al. 2007). Time delays in achieving those conditions were presumed to be significant, in the order of decades, even in the case that all nutrient reductions were made at once (Savchuck & Wulff 2007). **Table 4.1** summarizes the inputs to and outputs from the MARE/NEST calculations on maximum allowable loads to achieve 'good environmental status' and **Table 4.2** indicates the provisional nutrient reduction requirements of the countries that are based on the maximum allowable nutrient loads in **Table 4.1**.

Table 4.2 Provisional country-wise nutrient load reduction allocations, in tonnes per year (HELCOM 2007b).

	Phosphorus	Nitrogen
Denmark	16	17,210
Estonia	220	900
Finland	150	1,200
Germany	240	5,620
Latvia	300	2,560
Lithuania	880	11,750
Poland	8,760	62,400
Russia	2,500	6,970
Sweden	290	20,780
Transboundary pool	1,660	3,780
Sum	15,016	133,170

In addition to the BSAP, European directives such as the Marine Strategy Framework Directive (MSFD) and the Water Framework Directive (WFD) require the Baltic coastal countries that are EU Member States to reduce eutrophication to an acceptable level corresponding to good ecological/environmental status, thus giving further impetus to the implementation of the BSAP. The MSFD contains a eutrophication-related qualitative descriptor for determining good environmental status: '*Human-induced eutrophication is minimized, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters*' (MSFD Annex I). With this, the directive explicitly requires that eutrophication be specifically assessed and addressed in the implementation of the directive.

The current status of hypoxia and anoxia in the deep basins and coastal areas aggravates eutrophication and demands that we consider how to manage hypoxia in order to reduce eutrophication.

Hypoxia is largely due to the natural barrier created by the halocline, which inhibits mixing between the deeper water and the surface. In recent years, hypoxia in the Baltic Proper has been the worst ever with over half of the sea bottom and all water deeper than 80 m devoid of benthic animals and fish (Conley 2008). Hypoxia and anoxia are prevalent in coastal waters as well, and have an obvious link to increased primary production in such areas (Pitkänen et al. 2001). Even though the brackish nature of the Baltic Sea is the main factor to blame for poor ventilation of the deep basins, larger-scale climatic-oceanographic changes may also be associated with the less frequent intrusions of oxygen-rich saline water from the North Sea to the Baltic Sea (Heino et al. 2008). Owing to the linkage between hypoxia and eutrophication, possible management measures to increase oxygen in the deeps have been increasingly discussed and studied. Methods discussed vary from deepening of the Sound to increase the inflow to closing the Danish Straits to decrease the inflow of saline water, and thus eliminate stratification in the water column in order to provide active oxygenation of deep water. So far, these scenarios have been considered as unrealistic or likely to result in unfavourable outcomes. For the time being, only the idea of mixing the water around salinity stratification seems promising (Conley 2008). However, also new approaches such as oxygenation of water by replacing bottom vegetation could be being considered. Owing to the huge impact that the internal loading of nutrients can have on nutrient concentrations (e.g. Pitkänen et al. 2001), further research and small-scale pilot projects on possible management measures and their impacts should be encouraged.

Although the reduction of nutrient loads has been considered the key to reducing eutrophication of the Baltic Sea, there are recent claims that top-down effects in the food webs may play a more significant role in eutrophication than previously thought. On a global scale, fisheries has been considered the major human disturbance which usually precedes other human disturbances, including eutrophication, by making the ecosystem more vulnerable (Jackson et al. 2001). In a recent study from the Baltic Sea, high concentrations of summertime chlorophyll-*a* in the central Baltic were explained solely by top-down mechanisms, including a large population of zooplanktivorous sprat (*Sprattus sprattus*) and, consequently, a smaller biomass of grazing zooplankton (Casini et al. 2008). It was considered that sprat indirectly, via regulating zooplankton biomass, affects summer phytoplankton biomass. Sprat, on the other hand, was abundant due to the dramatic reduction of piscivorous cod (*Gadus morhua*), which has sprat as its main prey. There are current efforts to establish pilot projects to increase the quantities of piscivorous predator fish, at least in Sweden. In addition, targeted removal of sprat to reduce the sprat populations has been considered a potential means to enhance alleviation of eutrophication signals.

The agreed, currently implemented measures to combat eutrophication should be evaluated in the light of the projected environmental changes for the Baltic Sea region to be expected as a result of global climate change. An increase of the mean annual temperature by 3ºC to 5ºC has been projected for the Baltic Sea basin during this century (HELCOM 2007c). It is likely that the changing climate would also entail a general increase in annual precipitation, in particular, during the wintertime. Increased runoff, resulting from the increase in precipitation and reduced ice cover on northern rivers, would probably lead to increased nutrient loads from the drainage area to the Baltic Sea. Furthermore, changes in hydrographic conditions such as a decrease in average salinity would have an impact on convective mixing of the water and the distribution of nutrients. The increase in water temperature would stimulate primary production and increase bacterial activity. The World Wildlife Fund recently undertook a modelling study of the effects of climate change on eutrophication of the Baltic Sea and claimed that climate change will stimulate further eutrophication (WWF 2008). Nutrient load reductions of the magnitude identified in the HELCOM BSAP would compensate for the effects of climate change. Nevertheless, the effects of climate change would make the HELCOM strategic goal ‘Baltic Sea unaffected by eutrophication’ impossible to reach using the currently agreed reduction targets.

In 2005, shipping in the Baltic Sea contributed 9% (19 kt yr^{-1}) of a total of 208 kt yr^{-1} of airborne N deposited directly to the sea (Bartnicki 2007a). According to this estimate, shipping thus contributed ca. 2% of the total N load of 828 kt yr^{-1} to the Baltic Sea and 3.5% of the NOx load of 386 kt yr^{-1}. With a projected 2.6% annual traffic increase in the Baltic, and assuming no abatement measures, the estimated annual load from maritime traffic alone has been estimated to increase by roughly 50% until 2030 (Stipa et al. 2007).

Increased economic development, and thereby also increased pressures from human activity in the Baltic Sea region, will possibly contribute to an increase in eutrophication. Supplementary measures may be required to mitigate these negative environmental effects. Especially important are the developments taking place in the agricultural sector. It is highly likely that nutrient leaching from agriculture in the eastern part of the region will increase in the future owing to increased fertilizer use and increased livestock production (HELCOM & NEFCO 2007).

Eutrophication may also be further affected by demographic changes in the Baltic Sea region. Population increases, possibly coupled with increased wealth and consumption as well as a shift in tourism from southern Europe towards the north due to a changing climate, may put further pressure on the use of the sea and eutrophication.

On a positive note, nutrient loads from some point sources will be reduced owing to improvements in wastewater treatment and as a result of large projects and programmes that are being carried out, for example, in St. Petersburg, Kaliningrad and Poland.

The most important factor for reaching good ecological/environmental status with regard to eutrophication is political will, and cost-effective solutions must be available in order to motivate such political determination. Enhancing wastewater treatment to include chemical removal of phosphorus has been estimated as one of the most cost-efficient measures. Furthermore, substitution of phosphorus in laundry detergents is a measure already used in many European countries, as it is easy to employ and cost-effective. New paths should also be explored and collaboration between Baltic Sea countries could be enhanced by assessing the possibility of employing some degree of regional trading with nutrients (applying a mechanism similar to that of carbon trading). According to a recent report, trading has the potential to increase interest in nutrient removal even though there are legal issues to be solved and risks related to the large administrative needs of such a system (NEFCO 2008).

Despite the Baltic Sea often being praised as the most scientifically studied and best-documented sea region in the world, the quality of regional monitoring of the nutrient pollution entering the Baltic Sea is still relatively poor. HELCOM is the only body carrying out Baltic-wide assessments of nutrient loading to the sea and over the years HELCOM’s pollution load compilations have evolved and developed from a fragmented, methodologically incoherent approach (HELCOM 1980) to a coherent approach employing common guidelines (HELCOM 2004). Nevertheless, source apportionment of the nutrient loads is incomplete; in particular, the precision of estimating agricultural loads does not reflect the severity of the problem of agricultural nutrient inputs. In addition, pollution load compilations are produced far too infrequently. With the new compilation under way, there will be a time-lag of six years since the previous comprehensive compilation (PLC-4, HELCOM 2004).

The ecosystem approach to the management of human activities is a central feature of the BSAP and the MSFD also names it as one of its basic principles. The approach should be based on best available scientific knowledge of the ecosystem.

Despite frequent references to the Baltic Sea as one of the most studied seas in the world, the functioning of the ecosystem, especially complex cause-effect relationships, is not understood well enough to allow for the most cost-efficient combination of management measures. To develop an effective nutrient management strategy, an even better understanding of the Baltic Sea ecosystem is required. Aspects that would merit more in-depth knowledge are, for example, various lag times in the system, such as the leaching of nutrients from nutrient-rich soils or permanent burial of nutrients in the sediments, as well as relationships between nutrient concentrations, nutrient ratios, and eutrophication effects. More information is needed on food web interactions, for example, on the possible indirect effects of changes in fish or seal populations on eutrophication. Ecosystem thresholds and points-of-no-return, which are deeply symptomatic ecosystem states where management measures in the normal range no longer suffice, are also not well described or understood. Scientific understanding of the complex Baltic Sea ecosystem is imperfect, but according to the precautionary principle, which is a fundamental principle of the Helsinki Convention, preventive measures are to be taken also on the basis of incomplete knowledge.

In the HELCOM BSAP, the Contracting Parties have committed themselves to implementing adaptive management for restoring the good ecological/environmental status of the Baltic Sea and to the task of revisiting the nutrient reduction targets and measures of the BSAP, taking into account all new data and information. The Baltic Sea region is under multiple pressures, with increasing human activities and development together with ongoing climate change adding to the existing pressures. Although current measures may be reducing nutrient loads from some sources, increased loads can be expected from others. Adaptive management is a process of being flexible to evolving circumstances and learning by doing. The upcoming review of the BSAP nutrient reduction figures should incorporate all new data, information and scientific knowledge, and in the meantime, intensified research and monitoring should be carried out. Consequently, the means and measures to achieve good ecological and environmental status of the Baltic Sea with regard to eutrophication by the year 2021 should be periodically updated and corrected.

4.3 What are the costs?

Curbing eutrophication to a level of good ecological status has been estimated to require at least reductions to the total annual nutrient load by 135,000 t of nitrogen and 15,250 t of phosphorus (HELCOM 2007b). The reduction targets have been allocated to the different sub-basins of the Baltic and to the different countries. To reach the reduction targets, priority measures such as improvement of wastewater treatment, restrictions on the use of phosphorus in laundry detergents, measures to reduce nutrient leaching from agriculture, and actions to curb emissions of nitrogen to air especially from shipping were adopted in the Baltic Sea Action Plan. It was estimated that the improvement of municipal wastewater treatment alone would result in a reduction of the current phosphorus load by 6,700 t, covering over 40% of the needed reductions (HELCOM 2007b).

4.3.1 Cost-efficiency of measures for abating nutrient enrichment

The total costs of reaching the nutrient reduction targets are highly dependent on the cost-efficiency of the measures chosen. The cost-efficiency of measures varies depending on the type of measure and the conditions in the specific country or geographic location where the reductions are implemented (HELCOM & NEFCO 2007). Improvement of municipal wastewater treatment is a highly relevant measure, but its cost-effectiveness depends on population densities and current levels of wastewater treatment in the areas where improvements are implemented (HELCOM & NEFCO 2007). Within the agricultural sector, measures such as the use of catch crops and reduced fertilizer use are deemed as relatively cost-effective, while for shipping the reduction of nitrogen emissions by selective catalytic reduction seems to be a very cost-effective measure (HELCOM & NEFCO 2007). In addition to abatement measures at source, changes in land use to reduce the leaching of nutrients and the creation of nutrient sinks, such as wetlands and buffer zones to reduce the transport of nutrients to the coastal waters, are potentially cost-effective measures (Turner et al. 1999).

There are differences between the cost-efficiency of measures in different sectors and in different countries, and the BSAP stresses the importance of choosing cost-efficient measures. The cost-effectiveness of measures can be analysed by estimating the cost for each unit of reduced nutrient at source, i.e., a unit abatement cost (UAC), also taking into account the operational costs and technical lifetime (e.g., HELCOM & NEFCO 2007; NEFCO 2007). Measures with a UAC of less than €150,000 per tonne of reduced phosphorus were considered as cost-efficient and it is recommended that they should be implemented immediately (HELCOM 2007b). Analysis of a number of past projects in the Baltic Sea region showed that UACs for phosphorus reduction ranged from €20,000 to €1,900,000 per tonne of phosphorus reduced (NEFCO 2007). Projects on the eastern and southern side of the Baltic Sea were potentially ten times more cost-effective than projects carried out in the Nordic countries (NEFCO 2007). Similarly, Ollikainen & Honkatukia (2001) claimed that unit costs for the cheapest abatement investments are an order of magnitude lower in Estonia, Latvia, Lithuania, Poland and Russia than they are in the Nordic countries. These discrepancies in the cost-efficiency of abatement measures have also motivated an examination of prerequisites for nutrient trading schemes as a potential tool to enhance nutrient reductions (NEFCO 2008).

According to some authors, abatement measures for nitrogen are more costly than those for phosphorus (Gren et al. 1997; Turner et al. 1999). However, the situation may also be the opposite, as phosphorus is less mobile and therefore less easily abated (Karl Johan Lehtinen, pers. comm.). Turner et al. (1999) pointed out that phosphorus could serve as a keystone nutrient with the idea that when phosphorus is reduced, nitrogen is reduced simultaneously free of charge (Turner et al. 1999). This apparently does not apply the other way around, i.e., it is not true that when nitrogen is reduced then so is phosphorus.

Marginal costs of abatement measures increase with abatement effort, meaning that less reduction of a pollutant is obtained at higher cost as compared with earlier measures (Turner et al. 1999). Also, in the cases of municipal wastewater treatment, abatement measures within the agricultural sector and the reduction of nitrogen emissions from shipping, as mentioned above, the cost of measures per unit reductions achieved in all three sectors tends to increase along with effort and achieved load reductions (**Fig. 4.2**, NEFCO 2007).

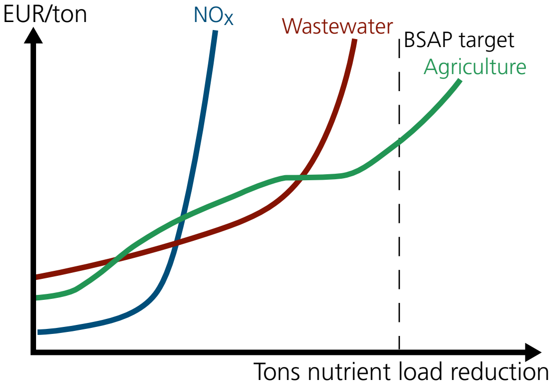


Figure 4.2: Illustrative curves of cost-effectiveness in three different sectors: nitrogen emission reductions in shipping (NOx), municipal wastewater treatment and agriculture in relation to the nutrient reduction targets of the Baltic Sea Action Plan (NEFCO 2007).

The cost-efficiency calculation of abatement measures at source does not take into account the impact of the reductions on the nutrient load to the coastal waters or on the status of nutrient concentrations in the sea. Particularly when a nutrient abatement measure takes place in the drainage area a long distance away from the coastal waters, the impact of the reductions in the sea is less than in the case of the same measure taken at a source discharging directly to the sea. This is due to the transformation and partial retention of nutrients on the way to the coastal waters (Wulff et al. 2001a). As a result, to have the desired impact in the coastal waters, a larger reduction at source is usually needed the longer the distance from the shore. Therefore, to give the same impact in the coastal waters, the marginal cost for reductions at source further inland is higher than the marginal cost for reduction at a source discharging directly into the sea (Turner et al. 1999).

Estimates of the impacts of nutrient reductions made in the drainage area on loading to coastal waters and on nutrient concentrations in the sea are based on many assumptions. Large natural variations in hydrology have an effect on the retention and transformation of waterborne nutrients. Furthermore, there are time lags between the actual reductions and their ecological responses, which pose a further challenge (Wulff et al. 2001a).

Estimations of impacts and the cost-efficiency of nutrient reductions are particularly sensitive to biological assumptions, e.g. leaching of nutrients from agricultural lands or the capacity of wetlands to remove nitrogen (Turner et al. 1999).

4.3.2 The total costs of curbing eutrophication of the Baltic Sea

Uniform solutions involving equal reduction percentages of nutrient loads in all countries are economically inefficient (Gren et al. 1997; Turner et al. 1999). A uniform 50% reduction of nitrogen loads from all coastal countries was estimated to be about €90 billion more costly than cost-effective reductions made with full international cooperation (Ollikainen & Honkatukia 2001). Only solutions which take into account the role of limiting nutrients in different sub-basins and rely on full international cooperation with redistributed abatement effort between the sub-basins and the countries are optimal (Turner et al. 1999). From the point of cost-effectiveness, an ultimate optimal solution would involve redistribution of effort so that equal marginal costs of abatement measures in all countries would be reached (Turner et al. 1999). Basically, nutrient trading schemes aim at achieving the same result via market mechanisms (NEFCO 2008).

The estimated total costs for reducing eutrophication of the Baltic Sea with a cost-effective solution have been estimated to range between €1,600 million and €16,500 million per year (**Table 4.3**). According to all examined studies, cost-effective solutions always involve a nutrient reduction model with full international cooperation where the measures are taken in countries and sectors yielding a maximum result with a minimum price.

4.3.3 The benefits from reducing eutrophication

To be able to estimate what is at stake, the costs of eutrophication abatement must be weighed against benefits to be gained from reducing eutrophication. The benefits consist of goods and services with both use and non-use values produced by the marine ecosystem. Nutrient recycling, water and climate regulation, production of fish and other food items, and recreational opportunities are among the ecosystem services provided by the Baltic Sea (Rönnbäck et al. 2007). Provision of these goods and services can be disturbed by exacerbating eutrophication or, conversely, their provision enhanced by reduced eutrophication.

On a global scale, marine ecosystems have been estimated to produce 63% of all the world's ecosystem services, with the total annual value of 33 trillion (10^{12}) US dollars (USD) (Costanza et al. 1997). The coastal ecosystems of the world produce services with an annual worth of USD 10.6 trillion. Nutrient cycling is by far the most important single service produced by marine ecosystems, with average annual global value of USD 17,075 per hectare. The Baltic Sea is among the most productive ecosystems, with much of the area providing services with an annual worth in the range of USD 2,000 to USD 3,000 per hectare (Costanza et al. 1997). In the case of the Baltic Sea, especially the nutrient assimilative capacity has been used free of charge and largely over-used (Turner et al. 1999).

Estimation of the benefits that can be attained from a marine environment or the improvement of its environmental status is challenging because most of the ecosystem's goods and services are not captured in commercial markets and have no price tag. One approach to estimating the value

Table 4.3 The costs, benefits and net benefits (in millions of euros per year) from combating eutrophication by reducing annual nutrient loads to the Baltic Sea with a cost-efficient solution, as estimated by various authors. na: not available.

Achieved nutrient reductions (per year)	Costs	Benefits	Net benefits	Reference
Cost-effective 50% reduction in total N and P loads	3,308	7,378	4,070	Turner et al. (1999)
Cost-effective 50% reduction in total N and P loads	3,308	3,356	48	Gren et al. (1997)
Cost-effective 50% reduction reduction of N and P loads	16,500	na	na	Ollikainen & Honkatukia (2001)
50% reduction of N and P	1,600	na	na	Wulff et al. (2001a)
Reductions of 100,000 N and 12,500 P	3,000	na	na	HELCOM & NEFCO (2007)

of a certain improvement or ecosystem property is to estimate individuals' willingness to pay by using, e.g., a contingent valuation method (Turner et al. 1999). To estimate benefits from reducing eutrophication, people's willingness to pay for an improvement in the marine environment was studied using a questionnaire (Gren et al. 1997; Turner et al. 1999). The benefits of a Baltic Sea with reduced eutrophication were estimated to be €7,378 million (69,310 million Swedish kronor) per year (Turner et al. 1999, **Table 4.3**). In another study, the Baltic basin-wide total benefits were estimated to be slightly more than €3,356 million (31,527 million Swedish kronor) per year (Gren et al. 1997). Owing to uncertainties and underlying assumptions in the methodology, the values were considered to be rough estimates.

The estimates of costs and benefits together indicate that there are net benefits that can be achieved from the reduction of eutrophication by curbing nutrient pollution. The estimates for net benefits range from €48 million to €4,070 million per year (**Table 4.2**). Despite the varying assumptions and uncertainties behind the estimates, the figures point to a policy message that reduction of eutrophication is economically beneficial and cost-efficient measures should be taken immediately.

4.3.4. Costs of non-action

The recent report on the economics of climate change (The Stern report) contained an estimate of economic costs that will result from climate change if no action is taken. Estimation of the costs resulting from continuation of business as usual was based on scientific reports on projected changes in the environment owing to increases in CO_2 levels and climate change, and corresponding impacts on the economy.

Estimation of the costs of inaction in combating eutrophication of the Baltic Sea should be based on 'business as usual' scenarios of future development of eutrophication and on estimating the impacts of possible future deterioration on the benefits derived from the Baltic Sea. Such Stern report-driven thinking on the marine environment is slowly evolving as some of the Baltic countries have undertaken to produce 'Baltic Sea Stern reports'. For the time being, there are only highly fragmentary ideas of what the costs of inaction could possibly be. HELCOM & NEFCO (2007) estimated that if eutrophication worsened to an extent where the commercial fisheries collapsed, the fish processing industry would suffer a loss of €4.5 billion, together with a loss of 50,000 jobs. According to the report, this would amount to only 0.2 % of the GDP in the region. It was also pointed out that this type of estimation focusing only on the resources and their market values does not fully cover their social values, which is also elucidated by higher values for reducing eutrophication derived using contingent valuation methods.

5 SYNTHESIS, CONCLUSIONS AND RECOMMENDATIONS

In this chapter, the linkages among the different eutrophication processes are summarized, reflecting the connections between nutrient loads and eutrophication status, indicating trends (**Chapter 5.1**). Based on this, scientific and action-related conclusions are drawn (**Chapter 5.2**), leading to recommendations (**Chapter 5.3**) and a future outlook (**Chapter 5.4**).

For many decades, the Baltic Sea has been affected by severe eutrophication. Elevated nutrient discharges cause more intense phytoplankton blooms, resulting in deterioration in the aquatic light climate, and thereby reducing the extent of submerged vegetation. Deep basins and long-lasting stratification enhance the accumulation of sinking phytoplankton biomass in bottom waters. In a region of poor water exchange, the result is severe oxygen depletion and the formation of abiotic zones. A particular problem, increasing the Baltic Sea's sensitivity to eutrophication, is a tendency for the development of toxic cyanobacteria blooms, which can have effects on the entire food chain.

This first HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea, covering the period 2001–2006, provides a Baltic Sea-wide overview of the effects of nutrient enrichment on eutrophication processes. The assessment links the effects of eutrophication to the causative factors such as nutrient enrichment and anthropogenic activities which result in emissions, discharges, and the losses and deposition of nutrients to the marine environment.

Tangible and cost-effective solutions to the problems of nutrient enrichment and eutrophication in the Baltic Sea must be informed by and based on a common understanding of how human activities, loads and effects are coupled. The problems are well-documented and the solutions are, in general, straightforward: nutrient emissions, discharges and losses as well as atmospheric deposition have to be reduced at source, thereby reducing nutrient loads to the Baltic Sea. This can only be achieved by changes in the way human activities are carried out in the catchment areas, e.g. land use, food and energy production, as well as by a significant reduction in nutrient losses from industrial and agricultural production and further improvements in wastewater treatment in the municipal and industrial sectors.

5.1 Synthesis of the assessment

The eutrophication status has been assessed and classified in 189 'areas' of the Baltic Sea, of which 17 are open areas and 172 are coastal areas, cf. **Fig. 5.1**. The open waters in the Bothnian Bay and in the Swedish parts of the northeastern Kattegat are classified as 'areas not affected by eutrophication'. It is commonly acknowledged that the open parts of the Bothnian Bay are close to pristine and that the northeastern Kattegat is influenced by Atlantic waters. Open waters of all other basins are classified as 'areas affected by eutrophication'. The fact that the open parts of the Bothnian Sea are classified as an 'area affected by eutrophication' is related to a well-documented increase in chlorophyll-*a* concentrations. For coastal waters, eleven have been classified as 'areas not affected by eutrophication' and 161 as 'areas affected by eutrophication'.

5.1.1 Nutrient loads to the Baltic Sea

The annual average waterborne loading of nitrogen to the Baltic Sea was estimated to be approximately 641,000 tonnes for the period 2001–2006. The average waterborne loading of phosphorus was estimated to be approximately 30,200 tonnes in the same period. Within the Baltic Sea catchment, there appears to be a slightly decreasing trend in the riverine and direct discharges of both nitrogen and phosphorus compared with the period 1995–2000. There are large variations in area-specific loading (cf. **Table 5.1**). The only part of the Baltic Sea with low specific loads of both nitrogen and phosphorus is the Gulf of Bothnia. Although these decreases are not yet reflected in reduced nutrient concentrations in the Baltic Sea, the results confirm the fact that the measures taken to reduce the nutrient load are effective. However, there is a time lag before a positive response to these actions can be observed in the receiving environments. Diffuse sources play a dominant role, especially for the nitrogen load, and with climate change there is a risk that future loads may increase again. Hydrological processes cannot be changed or controlled. Therefore, focus has to be on the implementation of load reduction measures to support further reductions in nutrient loading, especially in agriculture.

Atmospheric nitrogen deposition is assumed to be at least 25% of the total nitrogen input to the Baltic

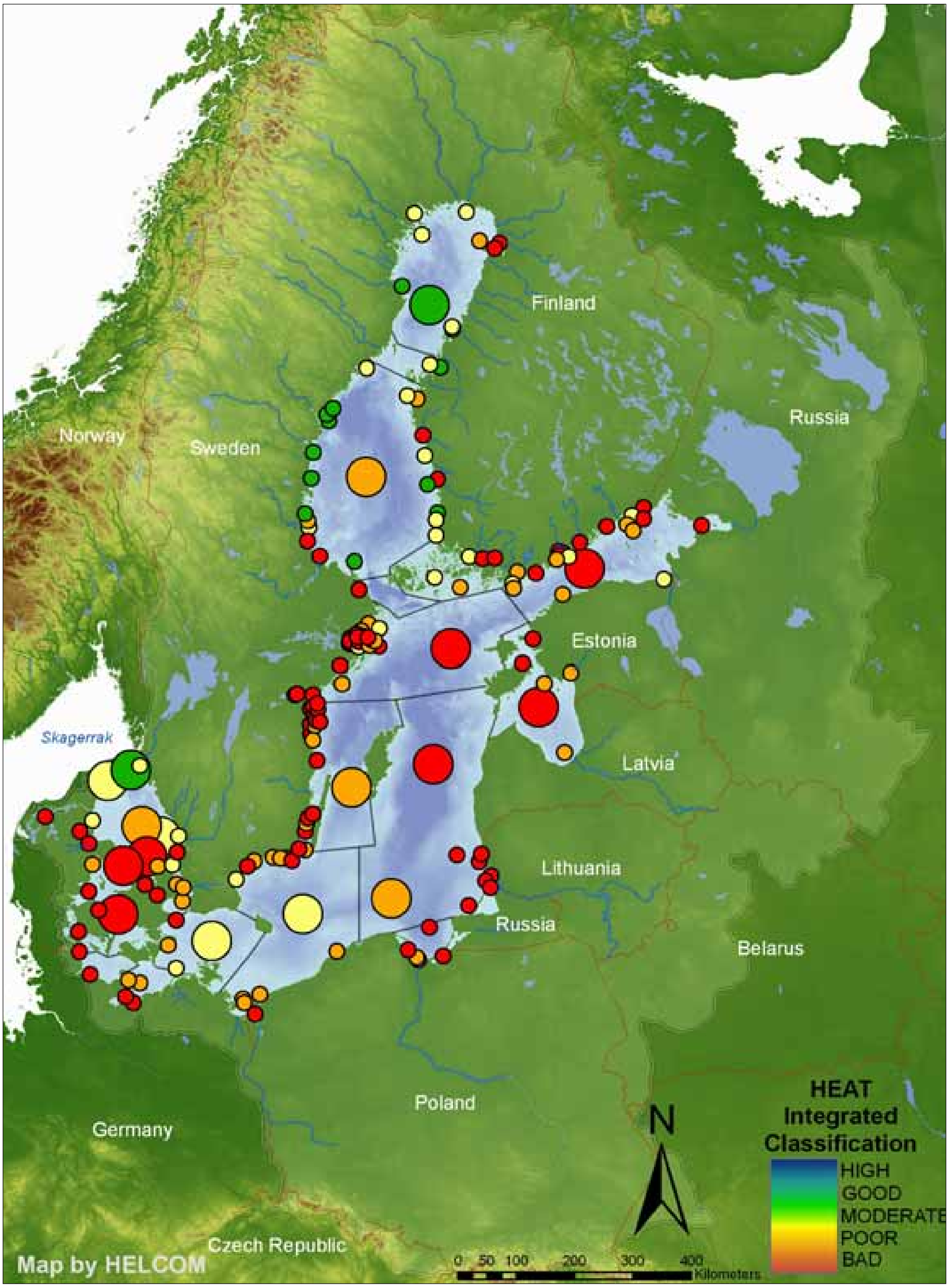


Figure 5.1 Integrated classification of eutrophication status based on 189 areas. Green = good status, yellow = moderate status, orange = poor status, and red = bad status. Good status is equivalent to 'areas not affected by eutrophication', while moderate, poor and bad are equivalent to 'areas affected by eutrophication'. Large circles represent open basins, while small circles represent coastal areas or stations. HEAT = HELCOM Eutrophication Assessment Tool.

Table 5.1 Waterborne inputs of nitrogen and phosphorus to the Baltic Sea in 2006.

	Area	TN	TN load/area	TP	TP load/area
		t	t km^{-2}	t	t km^{-2}
1	Gulf of Bothnia	109,069	0.94	4,612	0.04
2	Gulf of Finland	129,671	4.38	5,006	0.17
3	Gulf of Riga	58,417	3.58	2,659	0.16
4	Baltic Proper	227,838	1.03	12,875	0.06
5	Danish Straits	102,395	2.41	2,835	0.07
	Total	**627,390**	–	**27,987**	–

Sea. The total annual nitrogen deposition to the Baltic Sea was estimated at 196,000 tonnes in 2006. Nitrogen deposition to the Baltic Sea decreased by about 33% during the period 1980–2005. In future, it is assumed that nitrogen deposition will increase again owing to increased precipitation and growing contributions from shipping and agriculture.

5.1.2 Nutrients in the Baltic Sea

Once nutrients have been discharged, deposited, or lost to the marine environment, they can contribute to an increase in nutrient concentrations in seawater, also referred to as nutrient enrichment. Nutrient enrichment is not restricted to the water column. In some parts of the Baltic Sea, owing to widespread hypoxia, phosphorus is released from sediments, a process referred to as internal loading although the initial nutrient enrichment might have come from external, e.g., anthropogenic sources.

The Baltic Sea is one of the best-monitored marine ecosystems with some of the longest time series of data. Nutrient concentrations have been monitored for almost four decades, during which time the Baltic Sea has undergone major changes, involving significant nutrient enrichment and, more recently, decreasing nutrient levels in some regions. Owing to the differences in nutrient loads and physical characteristics of the different basins, responses in nutrient concentrations are different and common conclusions for the entire Baltic Sea can be difficult to draw. However, there are distinct spatial gradients for nutrients in the open waters, with nutrient levels increasing towards the Gulf of Finland and the Gulf of Riga, and phosphorus levels decreasing towards the Bothnian Bay. The Bothnian Bay and, to some extent also, the Bothnian Sea have similar nitrogen levels to the other regions, owing to loads from land and the atmosphere and the extensive phosphorus limitation of primary production. Nutrient levels generally decrease from the coast towards the open sea. This gradient is most pronounced in the Danish Straits and Baltic Proper. Nutrient concentrations in coastal areas of the Gulf of Finland are similar to those in the open sea owing to coastal-offshore mixing.

Nutrient concentrations increased up to the 1980s, and in all areas, except for the Gulf of Finland, phosphorus concentrations have declined during the past two decades. Nitrogen concentrations have declined in the Gulf of Riga, Baltic Proper and Danish Straits. These declines are partly caused by lower nutrient loads from land, particularly in the coastal zone, but changing volumes of hypoxia in the Baltic Proper significantly alter nutrient concentrations in bottom waters, and, through subsequently mixing, surface waters. This does not affect the Baltic Proper alone but also connecting basins through advective exchanges. In particular, the Gulf of Finland has been severely affected by internal loading of phosphorus from the sediments caused by poor oxygen conditions.

As all areas of the Baltic Sea experienced increasing nutrient concentrations up to the 1980s, symptoms of eutrophication became more apparent (**Fig. 5.2**). Management actions to reduce nutrient loads from land have shown results in some regions, reducing nutrient concentrations to the level of the 1970s. However, because the nutrient levels of the 1970s do not reflect acceptable elevated levels (e.g., levels less than 50% compared to reference conditions) and significant eutrophication effects are still observed in many areas, further reduction measures are generally necessary.

5.1.3 Eutrophication signals in the Baltic Sea

The growth of phytoplankton often responds directly to nutrient enrichment, but it is also potentially limited by low nutrient concentrations. In the Bothnian Bay, phytoplankton production is mostly P-limited, while in the Kattegat it is mostly N-limited. Variations in nutrient limitation patterns occur in relation to seasons, proximity to freshwater sources and during blooms of nitrogen-fixing cyanobacteria. Hence, nutrient management strategies need to address both nitrogen and phosphorus.

Phytoplankton biomass is widely monitored as chlorophyll-*a* in the Baltic Sea area. In most open and coastal Baltic areas, chlorophyll-*a* concentrations indicate the prevalence of eutrophication. In other words, EQR values derived for chlorophyll-*a* showed a clear deviation from reference conditions. In the open sea, the chlorophyll-*a* derived status was the highest in the Bothnian Bay and the Kattegat and lowest in the Gulf of Finland, the Northern Baltic Proper, and the Gulf of Riga. Typically, chlorophyll-*a* derived EQR values were lower in the inner coastal waters than in the outer coastal

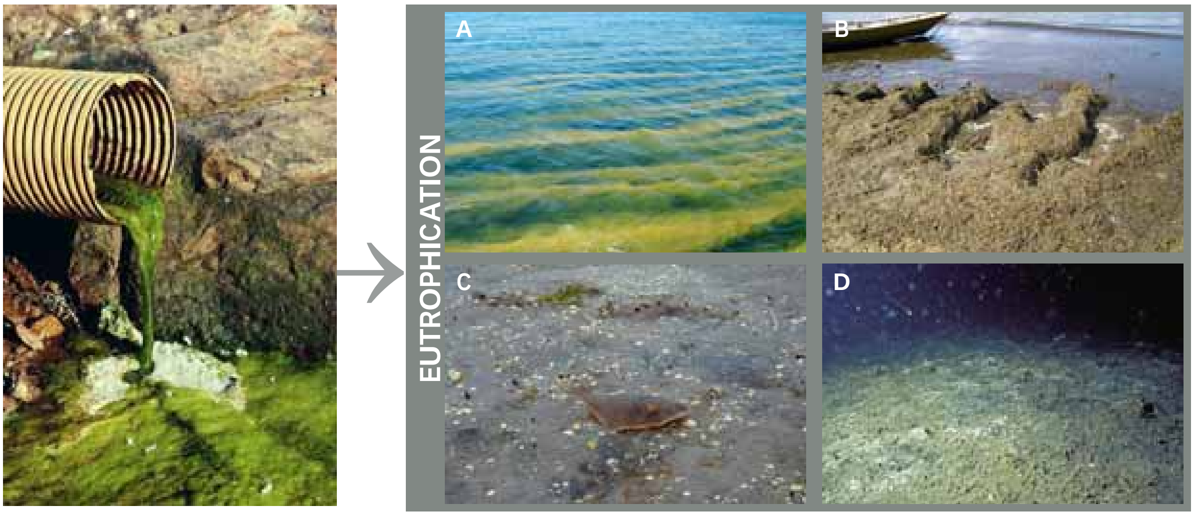


Figure 5.2 Increased loads lead to nutrient enrichment and eutrophication, e.g. blooms (A), loss of submerged aquatic vegetation (B), and in some cases even kills of fish and benthic invertebrates (C and D). Reductions of loads are prerequisites for trend reversal or recovery.

waters. During recent decades, chlorophyll-*a* concentrations have been increasing in most of the Baltic Sea sub-regions, although in the 2000s chlorophyll levels in many open sea areas showed signals of a decreasing tend.

The intensity and frequency of cyanobacteria blooms vary markedly, mainly owing to climatic variation, and no clear trends have been detectable during recent decades. Long-term data sets reveal that the phytoplankton species composition has changed during the past 100 years. These changes are linked to changes in nutrient levels, but also to other factors, such as climatic conditions and hydrography.

Reduced water transparency is an effect of increased loads and phytoplankton growth. Water transparency status has decreased in all Baltic Sea sub-areas, reflecting visible eutrophication effects in the entire Baltic Sea area, both at coastal and open-sea sites. Water transparency status is generally lower in the inner coastal and transitional waters and increases in the outer archipelagos and open sea. The status varies greatly among the sub-basins of the Baltic Sea, generally following the same geographical pattern in coastal and open-sea areas, and partly reflecting the regional differences in eutrophication status. It is the best in the Arkona Basin, acceptable in the Kattegat, Bornholm, Eastern and Western Gotland Basins and the Gulf of Riga, and significantly lowered in the Northern Baltic Proper, Gulf of Finland and the Gulf of Bothnia. A decrease in summer water transparency was observed in the open-sea areas in all Baltic Sea sub-regions over the past one hundred years. In the Kattegat, Arkona Basin, Bornholm Basin and Eastern Gotland Basin, the decreasing trend ceased during the past 15 to 25 years and since then the status has improved. In all other areas, the status is still deteriorating. Although the status is not acceptable in many areas, the recent trend reversal in southern sub-basins gives a positive signal of the possibilities of ecosystem recovery from the current eutrophic status.

Submerged aquatic vegetation (SAV) is an ecosystem component influenced by both open-sea conditions and terrestrial runoff. Generally, the level of eutrophication has caused serious changes in the Baltic Sea SAV communities, although the gaps in historic data do not allow us to identify the exact timing of larger shifts in communities. Present-day monitoring data show that in several areas the degradation of communities is ongoing. At the same time, positive signs of a slowing down or reversal of eutrophication effects on SAV parameters could be observed in areas of the northern Baltic Proper and Gulf of Finland, where the distribution of macrophyte species has recovered. In the Baltic Sea, a combination of salinity gradients, diversity of habitats and consequent differences in biotic interactions give rise to unique ecological conditions in each individual coastal area. Hence, similar species and communities may respond differently to similar stressors in different coastal areas. Recognized indicators therefore need to be adjusted to different areas, reflecting sub-regional variability in species or SAV community characteristics.

Oxygen concentration records in the Baltic extend back to the beginning of the 20th century. Data volumes were small until the 1970s, however, and some regions appear to remain under-sampled – in particular the Gulf of Bothnia and its associated sub-basins. The Gulf of Bothnia appeared to be free of both seasonal and long-term hypoxia, with the exception of some coastal sites. All other basins of the Baltic seem to have suffered from seasonal or permanent hypoxia during the assessment period. In 2002, large parts of the Baltic Sea, mainly in the southwestern and western parts, experienced the worst-ever hypoxia. In recent years, the occurrence of moderate hypoxia has increased in some areas. This is an indication of excessive oxygen consumption, most likely caused by eutrophication. Long-term hypoxic effects were observed in the Eastern and Western Gotland Basins, and the Northern Baltic Proper. This resulted from a combination of stagnation (caused by climatic factors) and ongoing eutrophication. The measurements indicate a significant increase in oxygen consumption since the 1960s at some stations in the Northern Baltic Proper, Sound and Kattegat. This is indicative of acute eutrophication.

The oxygen content of bottom waters is an important predictor of temporal changes in macrozoobenthic communities in open-sea areas. Quantification of eutrophication-induced hypoxia from naturally occurring oxygen deficiency is imperative for successful quantification of the eutrophication effects on benthic communities. Macrobenthic communities are severely degraded throughout the open-sea areas of the Baltic Proper and the Gulf of Finland, whereas conditions in the Arkona Basin, Danish Straits, open Kattegat and the Gulf of Bothnia, in general, are classified as being good. Macrozoobenthic communities in coastal waters are highly variable both between and within different sub-basins. In general, more sheltered and enclosed coastal water bodies are in a worse state than more exposed open coasts. The difficulty in defining historic reference conditions emphasizes the importance of conducting long-term monitoring over large spatial scales to be able to assess changes.

5.2 Conclusions

For practical reasons, the conclusions of the HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea have been divided into two categories: those relating to technical-scientific issues and those relating to action-oriented issues.

5.2.1 Technical and scientific conclusions

This assessment of eutrophication in the Baltic Sea covering the period 2001–2006 was made in 2006–2008 and was preceded by an activity in 2005 on the development of tools for the assessment of eutrophication.

The Baltic Sea countries share a joint understanding of the causes and effects of eutrophication as well as a long and close cooperation in regard to monitoring and assessment of eutrophication.

The commonly applied set of eutrophication-related indicators has an acceptable quality and represents a matching set of critical eutrophication signals. For most indicators, relatively long time series of data exist. The results and experiences from this eutrophication assessment, for example, with regard to the use of EQR values, provide a good basis for a revision of those indicators that are currently reported annually in HELCOM Indicator Fact Sheets. The work in regard to submerged aquatic vegetation and benthic invertebrate communities is of a standard that could serve as a basis for the development of new indicators.

This assessment represents a progression from a single-indicator based assessment of eutrophication status toward an integrated indicator-based assessment of eutrophication status. It uses the same indicators as the single-indicator approach, but applies a HELCOM Eutrophication Assessment Tool (HEAT) for an overall assessment and classification of the eutrophication status. HEAT distinguishes 'areas affected by eutrophication' from 'areas not affected by eutrophication' (see HELCOM (2009) for details). HEAT is indicator-based and makes use of synoptic information in regard to reference conditions, acceptable deviation from reference conditions, and actual environmental status. HEAT also makes use of the 'One out – All out principle' *sensu* the Water Framework Directive, which means that the overall classification of an assessed area is based on the most sensitive quality element. In addition, HEAT produces a provisional 'accuracy assessment' of the final classification results in order to assess the reliability of the final classification.

The assessment presents the current eutrophication or ecological status as an Ecological Quality Ratio, which is calculated on the basis of synoptic information on reference conditions and actual status, the latter for the period 2001–2006. One of the benefits of this approach is that it enables comparison between different parts of the Baltic Sea in a harmonized way and, hence, it exemplifies basin-, area- or site-specific deviations from unaffected conditions.

The use of the EQR approach is directly linked to the EU Water Framework Directive. Despite the advantage of the approach, it is sensitive to inaccurate information on reference conditions. Thus, it is crucial to further develop and improve the information base in regard to reference conditions. In general, reference conditions for chlorophyll-*a*, water transparency, submerged aquatic vegetation in coastal waters, and benthic invertebrate communities in open basins seem to be acceptable albeit with room for improvements. Reference conditions for nutrients, phytoplankton indicators other than chlorophyll-*a*, oxygen concentrations, and benthic invertebrate communities in coastal waters should been seen as a first but significant step towards more accurate information and consequently improved assessment on a Baltic Sea -wide scale.

In the assessment, reference conditions (RefCon) are considered to represent a status close to pristine conditions. However, regime shifts and/or shifting baselines can give reason to question the values currently used because there may be no return to the type of conditions that prevailed before and, instead, a good status in a new regime may represent a new kind of ecological condition. If a regime shift is irreversible or if the baseline is shifting, then there might be a need for a revision of the indicators, especially of reference conditions and acceptable deviations.

The acceptable deviations (AcDev) from reference conditions, used for setting boundaries between good environmental status in terms of eutrophication ('areas not affected by eutrophication') and unacceptable status ('areas affected by eutrophication'), originate from three processes: (1) a HELCOM project on the development of a tool for the assessment of eutrophication in the Baltic Sea (HELCOM 2006), (2) the implementation of the Water Framework Directive in coastal and transitional waters of the Baltic Sea including the intercalibration process, and (3) this assessment (details can be found in HELCOM 2009). The percentages for AcDev from reference conditions originating from (1) and (2) should be seen as a first attempt to establish reasonable boundaries between affected areas and non-affected areas. The AcDev percentages from (2) can in principle be used as they are, but only for coastal and transitional waters.

An important lesson is that there are three bottlenecks: the first is the spatial and temporal coverage of the monitoring network, the second is insufficient data on waterborne input figures, and the third is the synoptic data on RefCons to match the available monitoring data. There is an urgent need for improvement of these identified deficiencies.

The HELCOM COMBINE monitoring network in combination with national monitoring activities outside COMBINE provide good and scientifically well-justified data sources concerning the status of the marine environment for assessments such as this one. The spatial and temporal coverage is good compared to other regional seas in Europe. However, this assessment reveals the need for better geographical coverage, e.g. in the open parts of the Baltic Proper as well as in coastal waters of Latvia and Russia. The assessment is based mainly on indicators for nutrient concentrations, phytoplankton biomass (chlorophyll-*a*) and water transparency (Secchi depth). Indicators focusing on benthic communities are included, but only to a limited extent. There is clearly room for improvement and it is certain that the implementation of the WFD will lead to much better data sets on submerged aquatic vegetation and benthic invertebrate communities in coastal waters.

Based on the monitoring data for the period 2001–2006 and synoptic information on reference conditions, a total of thirteen areas in the Baltic Sea have been classified as ‘areas not affected by eutrophication’, of which two are open basins and eleven are coastal areas. Altogether 176 areas have been classified as ‘areas affected by eutrophication’, of which fifteen are open basins and 161 are coastal areas. **Fig. 5.3** shows individual classification results for the individual areas. **Fig. 5.4** shows the distribution of ‘eutrophication classes’ within fifteen major Baltic Sea basins.

The accuracy of the classification results is generally good. So-called confidence ratings were made of the data on which the 189 areas were assessed using a provisional accuracy assessment, under which 145 areas had a high or acceptable quality, cf. **Fig. 5.4**. For the 44 remaining areas, the provisional accuracy assessment showed that the quality of the classification should be improved, either by improving (1) the quality of information on reference conditions, (2) boundary setting for acceptable deviation, or (3) monitoring activities.

5.2.2 Action-oriented conclusions

Reductions in nutrient loadings have been achieved by most Baltic Sea countries; the long-term results are remarkable while the short-term development (2004–2006) is not as encouraging (see **Figs. 3.10** and **3.11**). The reductions have not yet resulted in a Baltic Sea unaffected by eutrophication. Hence, the good environmental status in terms of eutrophication as defined by the HELCOM Baltic Sea Action Plan (BSAP) has not yet been reached. Additional reductions and patience are needed; patience, in particular, because improvements in agricultural practices may need time before they take effect and result in lower loads.

The drivers that will result in a decrease in loads are, for the most part, proper implementation of national action plans and HELCOM recommendations as well as a number of legally binding international agreements and legislation such as the European directives addressing eutrophication. The most recent additions to the list of drivers

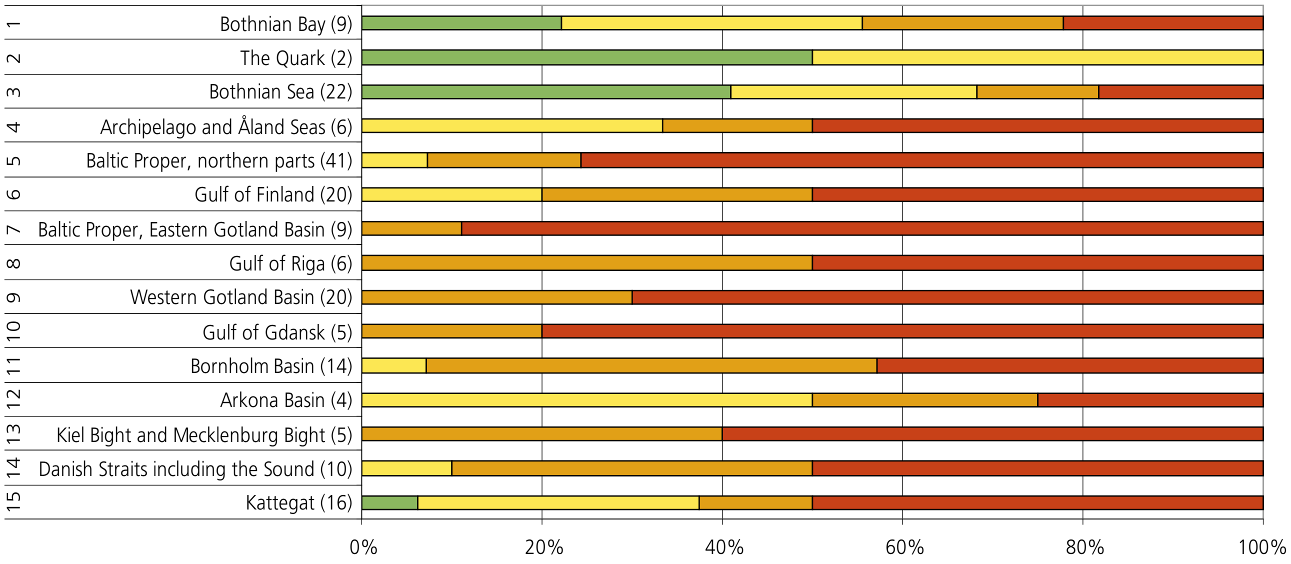
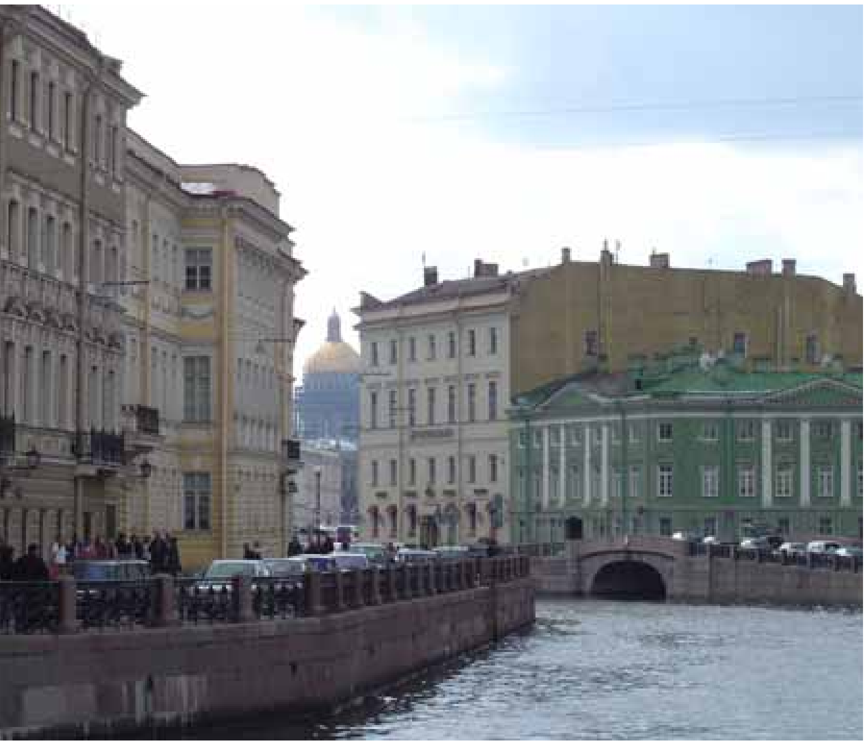


Figure 5.3 Overview of eutrophication classifications per basin based on the application of the HELCOM Eutrophication Assessment Tool (HEAT), see Annex 1 and HELCOM (2009) for details. The good class (green) equals ‘areas not affected by eutrophication’, while moderate, poor and bad classes (yellow, orange and red, respectively) equal ‘areas affected by eutrophication’.

Figure 5.4 Provisional 'accuracy assessment' of the eutrophication classifications. Coastal eutrophication classification results are presented per country (rows 1–9 in the upper panel) and basins (rows 1–15 in the lower panel). In the upper panel, open basins are presented separately (row 10). Classes I and II (light green and light yellow indicate a high or acceptable quality; class III (rose) indicates low quality. See HELCOM (2009) for details.

are the BSAP and the Marine Strategy Framework Directive. Implementation of the UWWTD, ND and WFD is essential, because tangible and durable improvements in the eutrophication status of the Baltic Sea rely on the load reductions provided via these directives and without their proper implementation, progress, if any, will be very slow or difficult to detect. Moreover, the implementation of these directives has already been taken into account when establishing the eutrophication segment and load reduction allocations of the BSAP.

The HELCOM BSAP and its eutrophication segment, adopted in November 2007, envisage provisional national load reductions tentatively set up on the basis of: (1) overall objectives and a set target for water transparency, (2) model calculations of maximum allowable loads and country-wise reduction targets, and (3) reduction scenarios and cost-efficiency (see **Chapter 4** and HELCOM 2007b). The approach employed is well-justified and well-documented and should be seen as an appropriate first step. The BSAP thus acknowledges that the figures related to targets and maximum allowable nutrient loads should be periodically reviewed and revised using a harmonized approach based on the most recent information and data. Further technical development of the modelling approach should be carried out by including a broader range of indicators, such as nutrient concentrations and chlorophyll-*a* concentrations in addition to the currently employed water transparency. In addition, greater coherence is needed between the modelling approach and the approach employed by this, and most likely also future, eutrophication assessments. Coherence could be enhanced by increasing the



Discharges from cities and industries have been significantly reduced, but more reductions are required to reduce eutrophication symptoms.

temporal resolution of the model to the level which is employed, *inter alia*, in this status assessment enabling a distinction between the different seasons instead of data averaged over the annual cycle. This would not only improve the reliability of the approach and load allocations, but also lead to greater credibility to the public, which has not yet been achieved by any other regional marine convention.

The total acceptable loads *sensu* the 2007 HELCOM Baltic Sea Action Plan and the 50% reduction target *sensu* the 1988 HELCOM Ministerial Declaration cannot be directly compared because there are slight differences in the approaches used. An indirect comparison indicates that the 2007 BSAP is stricter in terms of phosphorus than the 1988 Ministerial Declaration. In terms of nitrogen, however, it could appear that the 1988 Ministerial Declaration might be stricter. Nonetheless, this may not be significant for the following reasons: (1) the BSAP, addressing eutrophication using a holistic ecosystem approach, specifies a number of indicators with associated targets which are comparable with what would have been the ultimate effect of implementing the 50% reduction target, (2) the BSAP does not (yet) take a consequent implementation of the WFD into account in terms of expected load reductions, and (3) the BSAP will pursue declining loads and allow the Baltic Sea to recover from its present status.

5.3 Recommendations

The results and conclusions of the HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea lead to a suite of recommendations in regard to technical-scientific issues and action-oriented issues. Nonetheless, a first and important step regarding action-oriented recommendations is that the results of this assessment should be communicated to decision-makers in order to justify the need for higher political willingness to reduce eutrophication.

5.3.1 Technical and scientific recommendations

As an outcome of this assessment, a (better) quantitative definition of eutrophication is offered: ‘Eutrophication (noun) – is the enrichment of water by nutrients, especially nitrogen and/or phosphorus and organic matter, causing an accelerated growth of algae and higher forms of plant life including an increase in primary production and an unacceptable deviation in the structure, function and stability of aquatic communities present and in the quality of the water/ecosystem concerned, compared to reference conditions’.

Planning and coordinating assessments and linking them to other processes, e.g. revision of monitoring networks and Pollution Load Compilations, are necessary for an improved eutrophication assessment. This thematic eutrophication assessment should be seen as an initial step in the conduct of regular thematic and holistic assessments of eutrophication in the Baltic Sea. It is recommended that updates of this assessment be planned as soon at this assessment is published. A first step could be an update included in the HELCOM Holistic Assessment with an extension of the assessment period to 2001–2008. The next step could be to agree on a time frame for a second HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea, e.g. for

2009–2014 or any applicable period which would be in accordance with the subsequent reporting for the BSAP and also the WFD and MSFD. Thereby, duplicate work would be avoided and synergies gained.

Regarding the existing set of eutrophication indicators, it is recommended to continue their development and improvement and to make use of EQR values in the HELCOM Indicator Fact Sheets. Greater coherence between the eutrophication indicators used in indicator-based assessments, such as HEAT, and the HELCOM Indicator Fact Sheets should be sought. The result should be a set of eutrophication indicators, each one of which can serve partially on its own and have a Fact Sheet providing relevant background and policy-relevant information but also serve as part of the indicator set used in the assessment tool. New Indicator Fact Sheets focusing on submerged aquatic vegetation, benthic invertbrate communities, and overall eutrophication status are especially needed. By this arrangement, a regular updating of the overall eutrophication assessment would be facilitated.

It is recommended to continue to use Baltic Sea-wide integrated and indicator-based assessment tools, e.g. HEAT, when preparing the first holistic assessment of the status of and pressures on the marine environment of the Baltic Sea. It is also recommended to further develop and test such tools, especially regarding confidence rating and accuracy assessment. This would strengthen eutrophication monitoring and comparisons between different parts of the Baltic Sea, and also endorse and further justify investments in load reductions.

It also is recommended that improvements be made in the information regarding reference conditions, as the anchor of the indicator-based assessment, in terms of quality, consistency and geographical coverage. As a first step, it is recommended to improve and develop a Baltic Sea-wide harmonized ‘catalogue’ of reference conditions for nutrients, phytoplankton, water transparency, oxygen concentrations, and benthic macrophytes as well as invertebrate communities. Such a catalogue is a prerequisite for any updates of this assessment. Additionally, the further work on implementation of the Water Framework Directive on coastal and transitional waters, including intercalibration of methods, will to a large extent provide information on these water categories and should clearly be taken into account.

Furthermore, it is recommended that the principles for definition of an acceptable deviation from reference conditions, sometimes referred to as boundary setting, be harmonized, especially for open waters. This could imply initiation of research activities on natural variation and also a close coordination with the BSAP and the WFD and MSFD implementation processes.

Based on this assessment, it is recommended that existing monitoring networks be revised and subsequently improved. This assessment has not found any documentation or information that could justify reductions of monitoring activities. Special focus needs to be placed on: (1) the implication of the proposed quantitative definition of eutrophication, (2) better spatial coverage in coastal water, especially in Latvia and Russia, (3) better temporal coverage in the southern and eastern parts of the Baltic Proper, and (4) benthic communities, e.g. benthic invertebrate communities and submerged aquatic vegetation in coastal waters, especially in Finland, Germany, Latvia, Lithuania, Poland, and Russia.

It is recommended to strengthen the coherence of the work in relation to future Pollution Load Compilations and future assessments of eutrophication status for two reasons: firstly, to improve the quality and accuracy of assessments and, secondly, to improve the technical/scientific information on which management actions are being based. It may be worthwhile to re-organize the work of HELCOM’s subsidiary groups to match the requirements and structure of the BSAP.

A better and more action-oriented monitoring network, including modelling activities and remote sensing, will also lead to better assessments of the marine environment in the Baltic Sea. Following this recommendation will, on a short-term perspective, support the forthcoming HELCOM Holistic Assessment. On a longer perspective, the outcome will be a better scientific basis for management actions such as the implementation of the BSAP and European directives, in particular the WFD and MSFD.

5.3.2 Action-oriented recommendations

Because the majority of areas assessed were classified as 'areas affected by eutrophication', actions and measures need to be reconsidered and strengthened without delay: on a short-term perspective, to prevent further degradation and on a long-term perspective, to meet the objectives of the HELCOM Baltic Sea Action Plan, the WFD and MSFD.

Among those HELCOM countries that are also EU Member States, implementation of the UWWTD and ND and subsequent reduction of loads to the Baltic Sea are important in the process leading to a Baltic Sea unaffected by eutrophication. Both directives strive toward reductions of loads and nutrient enrichment with the ultimate aim of achieving waters not polluted by eutrophication. Because of the match between the goals of these directives and the BSAP, it is recommended that HELCOM annually takes stock of national progress, especially in regard to reductions in loading. This could be done under the BSAP implementation process. It is also recommended to use this assessment's classification of 'areas affected by eutrophication' as a tool for classifying 'sensitive waters' *sensu* the UWWTD and upstream 'vulnerable zones' *sensu* the ND.

'Areas not affected by eutrophication' *sensu* the HELCOM Baltic Sea Action Plan are identical to the overarching goals of the EU Member States under the WFD and MSFD. These directives aim to prevent further deterioration and strive towards good ecological/enviromental status. Interpreting this as an indirect prevention of increased loading, it is recommended that the HELCOM Contracting Parties report loading figures on an annual basis. Further, it is recommended that the BSAP implementation process - despite encouraging long-term trends in loading reductions - puts a strong focus on counteracting any short-term increases in loads.

Regarding the BSAP implementation process, it is also recommended to base the modelling scenarios and maximum load calculations on a broader range of indicators. As an initial and urgent step, it is recommended to include both nutrients and chlorophyll-*a* concentrations. This would improve both the reliability of the approach and the maximum load estimations, and lead to a public credibility not yet achieved by any other regional marine convention. Furthermore, it is recommended to develop a Baltic Sea -wide nutrient management strategy covering both open and coastal waters parallel to the implementation process of the BSAP and relevant EC directives. By doing so, the Baltic

Diffuse sources have been identified as a key target for management actions to abate eutrophication.

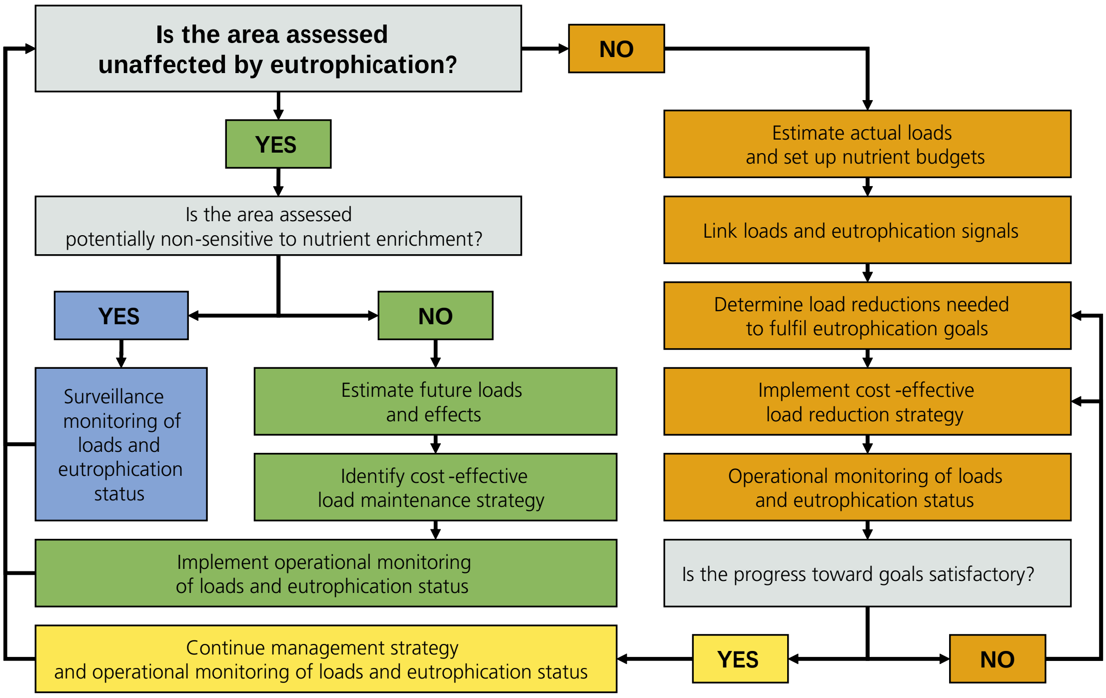


Figure 5.5 Suggested framework for a Baltic Sea-wide nutrient management strategy. Based on National Research Council (2000) and Backer (2008).

Sea countries would 'converge' implementation of the BSAP with implementation of other significant international instruments and make the most of the synergistic benefits. A tentative framework for a Baltic Sea-wide nutrient management strategy is outlined in **Fig. 5.5**.

5.4 Perspectives

Further development and strengthening of nutrient management strategies by the countries in the Baltic Sea catchment will be a result of multiple drivers, inspired by the BSAP, and often also national legislative plans implementing European directives and other national action. Which one is the most prominent or wide ranging is not an issue - the key is that loads are progressively reduced, especially in regard to diffuse sources. It should be clear that the eutrophication status will only improve if loads of both nitrogen and phosphorus are significantly further reduced. In this context, it should also be noted that there are strong links between eutrophication abatement and protection of marine biodiversity. Improving eutrophication status will, as a spin-off, result in significant improvements in habitat quality and conservation status in many parts of the Baltic Sea.

Possible shifting baselines and regime shifts pose a challenge. Management should address them by not permitting loads that give rise to irreversible regime shifts and also by setting allowable maximum loads *sensu* the HELCOM BSAP that permit the system to recover and develop to a status without eutrophication.

In addition to the work to reduce loads, climate change creates an extra challenge. Firstly, because precipitation is expected to increase especially in the northern part of the Baltic Sea catchment area; this may, in combination with increasing winter temperatures, lead to increased winter runoff and leaching of nutrients. Secondly, an increase in water temperatures will make benthic communities more vulnerable to eutrophication and hypoxia.

Ultimately, the effects of climate change would make the HELCOM strategic goal on eutrophication 'Baltic Sea unaffected by eutrophication' impossible to attain using currently agreed reduction targets. Further reductions are evidently required in order to reduce eutrophication effects, especially under a changing climate.

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GLOSSARY

Advection – is transport of substances in a fluid by the flow. An example of advection is the transport of pollutants in a river or the ocean by a current which carries these impurities along with it.

Algae – a large assemblage of lower plants, formerly regarded as a single group, but now usually classified in eight separate divisions or phyla, including the blue-green algae (*Cyanophyta*), green algae (*Chlorophyta*), brown algae (*Phaeophyta*), red algae (*Rhodophyta*), diatoms, and golden-brown algae (*Chrysophyta*). Marine macroalgae are commonly known as seaweeds.

Algal blooms – are usually naturally occurring algae that for some reason reach high enough concentrations to be a nuisance.

Anoxia – a state of oxygen depletion with lack of oxygen.

Aquatic – growing or living in or near water.

Atmospheric deposition – deposition of nutrients, heavy metals, and other pollutants from the atmosphere.

Autochtonous – originating or formed in the place where found.

Azoic – devoid of organic life.

BBI – the multimetric Brackish water Benthic Index (Perus et al. 2007) is used for classification of soft-bottom macrozoobenthic communities along the Finnish coastline.

Benthic – see benthos.

Benthos –organisms attached to, living on, in, or near the sea bed, river bed, or lake floor.

Biomass – the weight of organisms in a certain area either described with reference to volume or area.

Blue-green algae – marine and freshwater unicellular, colonial, or filamentous bacteria. Resemble algae in the way that they have chlorophyll pigments and can perform photosynthesis.

BQI – the Benthic Quality Index was developed in order to assess the status of Swedish coastal waters (Rosenberg *et al.* 2004, Blomqvist *et al.* 2006, Anon. 2008).

Catchment – the area of land which collects and transfers rainwater into a waterway.

Chlorophyll – any of several green pigments found in the chloroplasts of plants and in other photosynthesizing organisms. They mainly absorb red and violet-blue light energy for the chemical processes of photosynthesis.

Chlorophyll-*a* – a specific plant pigment essential for photosynthesis. It is quantitatively the most important pigment found in all photosynthetic phytoplankton cells.

Climate change – is the term used to describe changes in average climatic conditions over time periods ranging from decades to millions of years.

CORINAIR – atmospheric emissions inventory methodology.

Cyanobacteria – see blue-green algae.

Denitrification – a bacteria-mediated process in which nitrate is reduced to nitrogen gas.

Deposition – the dropping of material which has been picked up and transported by wind, water, or other processes.

Depth limit of macroalgae – maximum depth of findings of alive, attached specimen.

Diazotrophs – organisms which are able to use N_2 gas as nitrogen source for producing organic substances; see nitrogen fixation.

DIN – dissolved inorganic nitrogen. The sum of nitrate, nitrite, and ammonium, i.e., nitrogen forms that can be absorbed by plants.

DIP – dissolved inorganic phosphorus. The chemical form in which phosphorus can be absorbed by plants.

DKI – benthic quality can be assessed using the Danish multimetric index that measures species diversity and the sensitivity of the species, and gives these properties equal weight (Borja *et al.* 2007).

EC NEC Directive – European Commission National Emissions Ceilings Directive.

Eelgrass (*Zostera marina*) – a submerged flowering plant with dark green, long, narrow, ribbon-shaped leaves 20–50 cm in length with rounded tips that grows along the major part of the coasts of the Baltic Sea.

EMEP – Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe.

Epiphyte – a plant that grows on or is attached to another living plant.

EQR – Ecological Quality Ratio, being the ratio between reference conditions and actual ecological status.

EU – European Union.

Food chain – refers to direct links between organisms that describe how food energy is transferred through the ecosystem from the smallest primary producers to top predators. An example from the marine ecosystem is planktonic algae → copepods → fish → seal.

Global warming – is the impact on the climate from the additional heat retained due to the increased amounts of carbon dioxide and other greenhouse gases that humans have released into the earths atmosphere since the industrial revolution.

Gothenburg Protocol – Protocol to Abate Acidification, Eutrophication and Ground-level Ozone under the Convention on Long Range Transboundary Air Pollution for UN ECE.

Habitat-forming species – species that have the ability to modify the environment by their presence or functioning.

Halocline – strong, vertical salinity gradient that may cause vertical stratification.

HELCOM EUTRO – a HELCOM project focusing on development of tools for the assessment of eutrophication.

HELCOM EUTRO-PRO – a HELCOM project dealing with preparation, coordination and production of the HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea.

H_2S – hydrogen sulphide.

Humic – humic substances are a complex mixture of organic molecules that arise from the microbial degradation of dead plant and animal tissues. They are transported to the sea from land by river discharge. Humic substances are the most stable fraction of organic matter in soils and can persist for thousands of years. They are typically dark in colour and high concentrations can reduce water transparency. Humic substances are effective in binding positively charged ions (e.g. phosphorus ions in water environment) reducing their availability to organisms.

Hypoxia – a state of low oxygen values - see 'oxygen depletion' as well.

Macroalgae – plants that lack true roots, stems, leaves, or flowers. They mostly live attached to a hard substrate.

Kills – a kill is an unexpected and generally short-lived event marked by the conspicuous death of large numbers of fish (e.g. fish kill) or other organisms (e.g. benthic invertebrates).

Limiting nutrients – A limiting nutrient is a nutrient or trace element that is essential for plants to grow, but that is available in smaller quantities than are required by the plants and algae to increase in abundance. Therefore, if more of a limiting nutrient is added to an aquatic ecosystem, larger algal populations will develop until nutrient limitation or another environmental factor (e.g. light or water temperature) curtails production, although at a higher threshold than previously. It is often said that nitrogen is the limiting nutrient in marine and coastal waters; however, this general assumption is often incorrect. Phosphorus, carbon, silica and iron can also limit production in marine and coastal waters, and different trophic groups within the same ecosystem can be limited by different elements and nutrients.

Macroalgae – are an ancient class of large multicellular plants that resemble vascular plants but lack the complex array of tissues used for reproduction and water transport.

Macrophytes – all macroscopic plants in the aquatic environment.

Macrozoobenthos – animals larger than 1 mm living attached to, on, in, or near the sea bed, river bed, or lake floor.

MarBIT – MarBIT is a multi-metric assessment system to rate the biological quality of macrozoobenthos communities in the German part of the Baltic Sea (Anon. 2006).

Marine – of, or pertaining to, the sea, the continuous body of water covering most of the earth's surface and surrounding its land masses. Marine waters may be fully saline, brackish, or almost fresh.

Monitoring – is regular gathering of information, and the preliminary analysis of this information, in order for day-to-day management or evaluation.

MSFD – Marine Strategy Framework Directive.

Nitrate (NO_3) – an important nitrogen-containing nutrient. The chemical form in which plants take up most of their nitrogen. It is the salt of nitric acid.

Nitrogen (N) – a chemical element that constitutes about 80% of the atmosphere by volume. Nitrogen is an important part of proteins and is essential to living organisms.

Nitrogen fixation – conversion of N_2 gas to a form that is available for use by organisms.

Nutrient – a chemical element which is involved in the construction of living tissue that is needed by both plants and animals. The most important in terms of amount are carbon, hydrogen, and oxygen, with other essential elements including nitrogen, potassium, calcium, sulphur, and phosphorus.

Opportunistic algae – algae species that take advantage of a wide range of resources, habitats or environmental conditions. Opportunistic species have highly flexible life needs and usually very short life cycle, high productivity and rapid growth.

Organic material – once-living material (typically with high carbon content), mostly of plant origin.

Organism – an individual form of life. An animal, a plant, or a bacterium.

Oxygen – a non-metallic element constituting 21 percent of the atmosphere by volume. Oxygen is produced by autotrophic organisms and is vital to oxygen-breathing organisms.

Oxygen depletion – a situation where the demand for oxygen has exceeded its supply, leading to low concentrations of oxygen. Low oxygen concentrations are normally found in the water close to the sea bottom. In the Baltic Sea area, concentrations below 4 mg O_2 per litre are defined as oxygen depletion and concentrations below 2 mg O_2 per litre are defined as severe, acute oxygen depletion.

Pelagic – living and feeding in the open sea; associated with the surface or middle depths of a body of water; free swimming in open waters, not in association with the bottom.

Phosphate (PO_4) – an important phosphorus-containing nutrient. It is the chemical form in which plants take up phosphorus.

Phosphorus (P) – a non-metallic chemical element.

Photosynthesis – primary production, carbon production or simply 'production' is the process whereby pigments such as chlorophyll-*a* in plants and algae capture sunlight and covert it to organic matter and oxygen. Plankton generally has a Redfield molar proportion (e.g. $C_{106}H_{263}O_{110}N_{16}P_1$). Therefore photosynthesis by phytoplankton can be represented by the following reaction:

$$106\ CO_2 + 16\ NO_3^- + HPO_4^{2-} + 122\ H_2O + 18\ H^+ = C_{106}H_{263}O_{110}N_{16}P_1 + 138\ O_2$$

Phytoplankton – the plant plankton and primary producers (i.e., drifting, more or less microscopic, photosynthetic organisms) of aquatic ecosystems.

Plankton – free, passively floating organisms (animals, plants, or microbes) in aquatic systems.

Primary production – the production by autotrophs.

psu – practical salinity unit (almost equivalent to parts per thousand or ‰).

Pycnocline – a layer in the water in which water density increases rapidly with depth.

Redfield ratio – the 'Redfield ratio' or 'Redfield stoichiometry' refers to the molar ratio of carbon (C), nitrogen (N) and phosphorus (P) in phytoplankton (principally diatoms). When nutrients are not limiting, most phytoplankton has the following molar ratio of elements: C:N:P = 106:16:1.

Salinity – is the mass fraction of salts in water.

SAV – see submerged aquatic vegetation.

Secchi depth – a measure of the clarity of the water.

Seagrass – marine flowering plants, which generally inhabit soft substrates and attach to the bottom with roots.

Silt – fine sand, clay, or other material carried by running water and deposited as a sediment, especially in a channel or harbour.

Spring bloom intensity index – the index takes into account the chlorophyll-*a* concentrations and the duration of the bloom, and integrates this information into a single index value (Fleming & Kaitala, 2006a). The beginning and the end of the bloom are defined by a chlorophyll-*a* threshold level of 5 µg/l. The index value increases with increasing spring bloom intensity.

Stratification – physical layering of the water column resulting from density differences caused by salinity or temperature variation.

Submerged aquatic vegetation – aquatic vegetation such as seagrasses and seaweeds, that cannot withstand excessive drying and therefore live with their leaves at or below the water surface. Submerged aquatic vegetation provides an important habitat for numbers of other aquatic organisms.

TN – total nitrogen which includes dissolved inorganic and organic nitrogen and organically bound nitrogen

Tot-N – see TN.

Tot-P – see TP.

TP – total phosphorus which includes dissolved inorganic and organic phosphorus and organically bound phosphorus.

UN ECE – United Nations Economic Commission for Europe.

Upwelling – the rise of sea water from depths to the surface, typically bringing nutrients to the surface.

WFD – Water Framework Directive.

Zooplankton – small planktonic animals in fresh or sea water with almost none or no swimming capacity. They are, therefore, transported randomly by water movements.

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ANNEX 1: CLASSIFICATION OF EUTROPHICATION STATUS

This annex briefly describes the HELCOM Eutrophication Assessment Tool (HEAT) and summarizes the results of the HEAT classifications presented in Chapter 5 (**Figs. 5.1**, **5.3** and **5.4**).

In the Baltic Sea Action Plan, the Contracting Parties acknowledged that a harmonized approach to assessing the eutrophication status of the Baltic Sea is required. Therefore, the Contracting Parties agreed to further develop a common HELCOM assessment tool for use in a Baltic-wide thematic assessment of eutrophication in coastal as well as open sea waters.

HEAT is a multi-metric indicator-based tool for assessment of eutrophication status. HEAT has been developed specifically for the HELCOM Integrated Thematic Assessment of Eutrophication in the Baltic Sea.

Ecological objectives related to eutrophication were adopted in the HELCOM Baltic Sea Action Plan. They are: concentrations of nutrients close to natural levels, clear water, natural level of algal blooms, natural distribution and occurrence of plants and animals, and natural oxygen levels.

HEAT is based on existing indicators, which for this purpose have been grouped as follows: (1) physical-chemical features (PC), (2) phytoplankton (PP), (3) submerged aquatic vegetation (SAV), and (4) benthic invertebrate communities (BIC). Groups 1 and 2 (PC and PP) are considered 'primary signals' of eutrophication, while groups 3 and 4 (SAV and BIC) are considered 'secondary signals'. Within the four mentioned groups, HEAT allows weighting between indicators. Hence, indicators thought to be very good can be given a higher weight than an indicator with a low quality and vice versa.

For each individual indicator, an interim classification is made. The classification system has five classes: high and good correspond to 'areas not affected by eutrophication' and moderate, poor and bad, which correspond to 'areas affected by eutrophication'. Details of the classifications including a description of methodology, overview of the indicators used, reference conditions (RefCon), acceptable deviations (AcDev) and actual status (AcStat 2001–2006) can be found in the Background Report (HELCOM 2009b).

All countries except Denmark have used 2001–2006. Denmark has used the period 2001–2005 for the Kattegat and Great Belt and 2001–2004 for all other areas (areas in the Arkona Basin, the Sound, and Little Belt).

Indicators are combined within groups (quality elements) and ultimately combined into an assessment of 'overall eutrophication status'. This final step makes use of the 'One out – All out' principles *sensu* the Water Framework Directive. This implies that the overall classification of eutrophication status is based on the most sensitive quality element' similar to the WFD.

In some coastal areas, the classification presented in the Baltic Sea-wide eutrophication assessment cannot be directly compared to the results of national assessments and the Baltic Sea intercalibration exercise *sensu* the WFD owing to differences in spatial and temporal scaling, as well as the use of parameters that are considered supporting in WFD.

As a precautionary remark, it should be emphasized that the classifications make use of slightly different 'assessment units', e.g. open basins, areas/waterbodies as well as single stations. For clarity, open basins are shown in capitals (as BASINS), areas and waterbodies are indicated with regular font style, while single stations are indicated in *italics* (as *Stations*).

Fig. A1.1 demonstrates an example of HEAT calculations from Odense Fjord, Denmark. Information regarding indicators used, reference conditions, acceptable deviation, actual status, weighting between indicators and classification of overall eutrophication status is shown. Confidence rating and accuracy assessment are also shown. The same detailed information in regard to all other areas assessed can be found in HELCOM (2009).

Tables A1.1 and **A1.2** present a summary of the Annexes 1.1, 1.2, 1.3, 1.4, 1.5, 1.6, 1.7, 1.8, 1.9, 1.10, 1.11, 1.12, 1.13, 1.14 and 1.15. **Table A1.3** presents a summary of the provisional assessment of the accuracy of the eutrophication classification produced by HEAT.

Table A1.1: Distribution of the 189 'assessment units' between countries and basins.

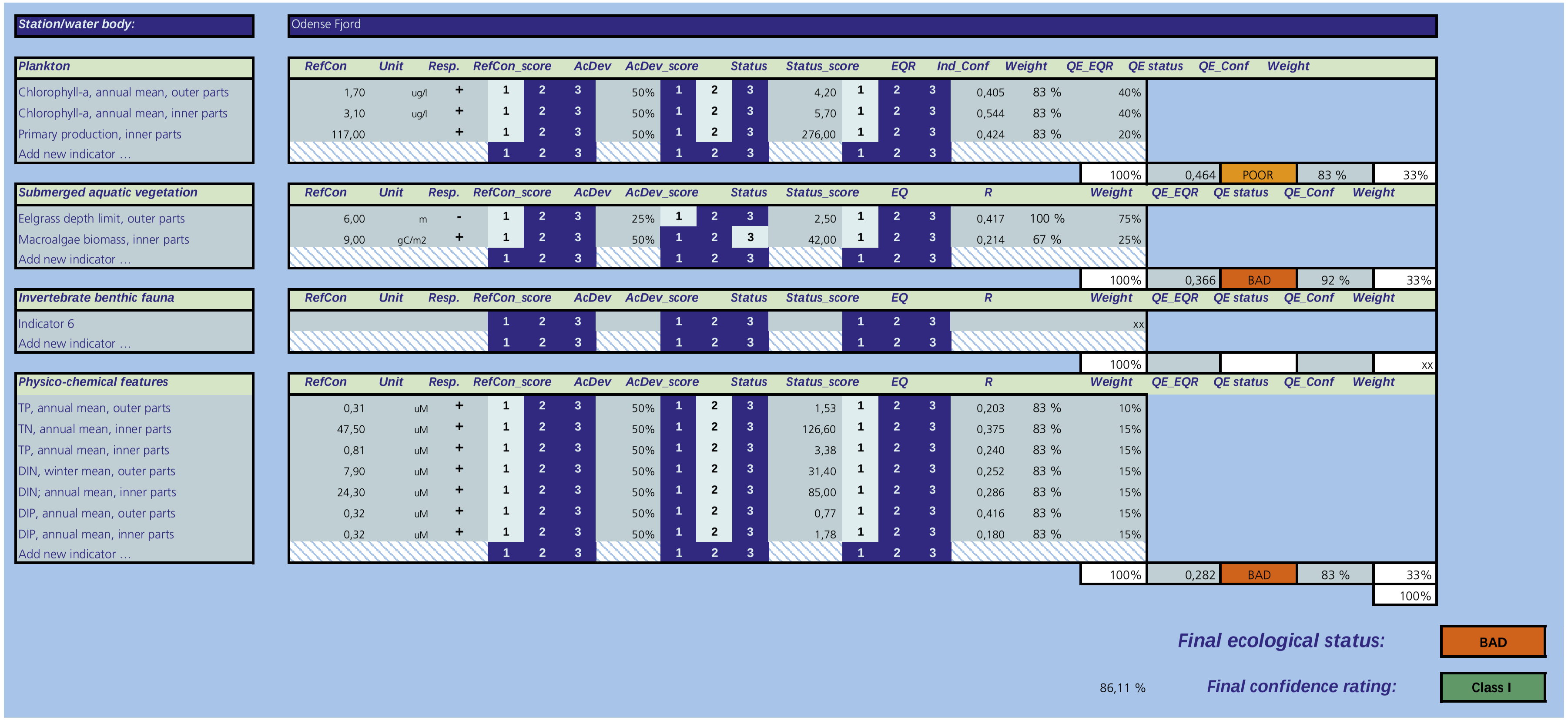
Basin	**Country**										**Sum**
	DEN	***EST***	***FIN***	***GER***	***LAT***	***LIT***	***POL***	***RUS***	***SWE***	***Open***	
1. Bothnian Bay	-	-	6	-	-	-	-	-	2	1	**9**
2. The Quark	-	-	2	-	-	-	-	-	-	-	**2**
3. Bothnian Sea	-	-	9	-	-	-	-	-	12	1	**22**
4. The Archipelago and Åland Seas	-	-	5	-	-	-	-	-	1	-	**6**
5. Baltic Proper, northern parts	-	-	-	-	-	-	-	-	40	1	**41**
6. Gulf of Finland	-	2	16	-	-	-	-	1	-	1	**20**
7. Baltic proper, Eastern Gotland Basin	-	-	-	-	-	-	-	1	-	2	**9**
8. Gulf of Riga	-	4	-	-	1	-	-	-	-	1	**6**
9. Western Gotland Basin	-	-	-	-	-	-	-	-	19	1	**20**
10. Gulf of Gdansk	-	-	-	-	-	-	5	-	-	-	**5**
11. Bornholm Basin	-	-	-	1	-	-	4	-	8	1	**14**
12. Arkona Basin	2	-	-	1	-	-	-	-	-	1	**4**
13. Kiel Bight and Mecklenburg Bight	-	-	-	5	-	-	-	-	-	-	**5**
14. Danish Straits including the Sound	5	-	-	1	-	-	-	-	3	1	**10**
15. Kattegat	8	-	-	-	-	-	-	-	3	5	**16**
Sum	**5**	**6**	**38**	**8**	**1**	**6**	**9**	**2**	**88**	**16**	**189**

Table A1.2: Overall eutrophication status in 15 Baltic Sea basins. Please observe that Fig. 5.3 is based on these classification results.

Basin	**Unaffected Areas**		**Affected Areas**			**Sum**
	High	***Good***	***Moderate***	***Poor***	***Bad***	
1. Bothnian Bay	0	2	3	2	2	**9**
2. The Quark	0	1	1	0	0	**2**
3. Bothnian Sea	0	9	6	3	4	**22**
4. The Archipelago and Åland Seas	0	0	2	1	3	**6**
5. Baltic Proper, northern parts	0	0	3	7	31	**41**
6. Gulf of Finland	0	0	4	6	10	**20**
7. Baltic proper, Eastern Gotland Basin	0	0	0	1	8	**9**
8. Gulf of Riga	0	0	0	3	3	**6**
9. Western Gotland Basin	0	0	0	6	14	**20**
10. Gulf of Gdansk	0	0	0	1	4	**5**
11. Bornholm Basin	0	0	1	7	6	**14**
12. Arkona Basin	0	0	2	1	1	**4**
13. Kiel Bight and Mecklenburg Bight	0	0	0	2	3	**5**
14. Danish Straits including the Sound	0	0	1	4	5	**10**
15. Kattegat	0	1	5	2	8	**16**
Sum	**0**	**13**	**28**	**46**	**102**	**189**

Table A1.3: Overview of the provisional accuracy assessment for each basin. Please observe that Fig. 5.4 is based on these provisional results.

Basin	**Class I**	**Class II**	**Class III**	**Sum**
1. Bothnian Bay	0	5	4	**9**
2. The Quark	0	1	1	**2**
3. Bothnian Sea	0	9	13	**22**
4. The Archipelago and Åland Seas	0	5	1	**6**
5. Baltic Proper, northern parts	1	29	11	**41**
6. Gulf of Finland	0	17	3	**20**
7. Baltic proper, Eastern Gotland Basin	1	7	1	**9**
8. Gulf of Riga	0	5	1	**6**
9. Western Gotland Basin	1	16	3	**20**
10. Gulf of Gdansk	1	2	2	**5**
11. Bornholm Basin	0	12	2	**14**
12. Arkona Basin	0	4	0	**4**
13. Kiel Bight and Mecklenburg Bight	0	5	0	**5**
14. Danish Straits including the Sound	6	4	0	**10**
15. Kattegat	5	10	1	**16**
Sum	**15**	**131**	**43**	**189**



Station/water body:	Odense Fjord																				
Plankton	RefCon	Unit	Resp.	RefCon_score			AcDev	AcDev_score			Status	Status_score			EQR	Ind_Conf	Weight	QE_EQR	QE status	QE_Conf	Weight
Chlorophyll-a, annual mean, outer parts	1,70	ug/l	+	1	2	3	50%	1	2	3	4,20	1	2	3	0,405	83 %	40%				
Chlorophyll-a, annual mean, inner parts	3,10	ug/l	+	1	2	3	50%	1	2	3	5,70	1	2	3	0,544	83 %	40%				
Primary production, inner parts	117,00		+	1	2	3	50%	1	2	3	276,00	1	2	3	0,424	83 %	20%				
Add new indicator …				1	2	3		1	2	3		1	2	3							
																	100%	0,464	POOR	83 %	33%

Submerged aquatic vegetation	RefCon	Unit	Resp.	RefCon_score			AcDev	AcDev_score			Status	Status_score			EQ	R	Weight	QE_EQR	QE status	QE_Conf	Weight
Eelgrass depth limit, outer parts	6,00	m	-	1	2	3	25%	1	2	3	2,50	1	2	3	0,417	100 %	75%				
Macroalgae biomass, inner parts	9,00	gC/m2	+	1	2	3	50%	1	2	3	42,00	1	2	3	0,214	67 %	25%				
Add new indicator …				1	2	3		1	2	3		1	2	3							
																	100%	0,366	BAD	92 %	33%

Invertebrate benthic fauna	RefCon	Unit	Resp.	RefCon_score			AcDev	AcDev_score			Status	Status_score			EQ	R	Weight	QE_EQR	QE status	QE_Conf	Weight
Indicator 6				1	2	3		1	2	3		1	2	3			xx				
Add new indicator …				1	2	3		1	2	3		1	2	3							
																	100%				xx

Physico-chemical features	RefCon	Unit	Resp.	RefCon_score			AcDev	AcDev_score			Status	Status_score			EQ	R	Weight	QE_EQR	QE status	QE_Conf	Weight
TP, annual mean, outer parts	0,31	uM	+	1	2	3	50%	1	2	3	1,53	1	2	3	0,203	83 %	10%				
TN, annual mean, inner parts	47,50	uM	+	1	2	3	50%	1	2	3	126,60	1	2	3	0,375	83 %	15%				
TP, annual mean, inner parts	0,81	uM	+	1	2	3	50%	1	2	3	3,38	1	2	3	0,240	83 %	15%				
DIN, winter mean, outer parts	7,90	uM	+	1	2	3	50%	1	2	3	31,40	1	2	3	0,252	83 %	15%				
DIN; annual mean, inner parts	24,30	uM	+	1	2	3	50%	1	2	3	85,00	1	2	3	0,286	83 %	15%				
DIP, annual mean, outer parts	0,32	uM	+	1	2	3	50%	1	2	3	0,77	1	2	3	0,416	83 %	15%				
DIP, annual mean, inner parts	0,32	uM	+	1	2	3	50%	1	2	3	1,78	1	2	3	0,180	83 %	15%				
Add new indicator …				1	2	3		1	2	3		1	2	3							
																	100%	0,282	BAD	83 %	33%
																					100%

	Final ecological status:	BAD
86,11 %	*Final confidence rating:*	Class I

Figure A1.1: HEAT calculations from Odense Fjord, Denmark including both inner and outer parts of the fjord. 'RefCon' = reference conditions; 'Resp' indicates either a numerically positive response (+) or a numerically negative response (÷); 'RefCon score' rates the quality of the RefCon value (1 = high, 2 = OK, 3 = poor); 'AcDec' = acceptable deviation from RefCon; 'AcDev_score' rates the quality of the AcDev value; 'Status' = actual status 2001-2006; EQR = ecological quality ratio calculated per indicator; 'Ind_Conf' indicates the accummulated quality of the specific indicator; ' Weight' shows the weight given to the indicators within a group; 'QE_EQR' = ecological quality ration per group of indicators; QE_status' = the classification of the group of indicators; 'QE_Conf' indicates the quality of the assessment/classification per group of indicators.

Annex 1.1: Bothnian Bay (9)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
1. BOTHNIAN BAY	0.729	0.668	–	0.830	**GOOD**
2. *Bothnian Bay, outer, LAV-4*	0.646	0.687	–	0.561	**MODERATE**
3. *Bothnian Bay, outer, Kokkola*	0.541	0.576	–	–	**MODERATE**
4. *Bothnian Bay, outer, Hailuoto*	0.788	0.395	–	–	**POOR**
5. *Bothnian Bay, inner, Luodonselkä*	0.373	0.400	–	–	**BAD**
6. *Bothnian Bay, inner, Kolmikulma*	0.768	0.360	–	0.836	**BAD**
7. *Bothnian Bay, inner, Kokkola*	0.545	0.857	–	–	**POOR**
8. Central Bottenviken Coast & F9	0.904	–	–	0.173	**GOOD**
9. Gussöfjärden	1.000	–	–	0.023	**MODERATE**
🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	☐ No or insufficient data

Annex 1.2: The Quark Area (2)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
10. *The Quark, outer, Vavy-19 Storbådan*	0.699	0.596	–	0.855	**MODERATE**
11. *The Quark, inner, Vav-3*	0.787	0.698	–	–	**GOOD**
🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	☐ No or insufficient data

Annex 1.3: Bothnian Sea (22)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
12. BOTHNIAN SEA	0.724	0.508	–	0.834	**POOR**
13. *Bothnian Sea, outer, Pran, Truutin Pauha*	0.564	0.761	–	–	**MODERATE**
14. *Bothnian Sea, outer, Pome 280*	0.628	0.778	–	–	**GOOD**
15. *Bothnian Sea, outer, Domarklobban*	0.628	0.636	–	–	**MODERATE**
16. *Bothnian Sea, inner, Vav-14*	0.586	0.368	–	0.436	**BAD**
17. *Bothnian Sea, inner, Uusikaupunki*	0.636	1.000	–	–	**MODERATE**
18. *Bothnian Sea, inner, Rauma Rounakari*	0.814	1.000	–	0.753	**GOOD**
19. *Bothnian Sea, inner, Pome 64*	0.355	0.145	–	–	**BAD**
20. *The Quark, outer, Vav-11*	0.558	0.610	–	–	**MODERATE**
21. *The Quark, inner, Vav-9*	0.548	0.440	–	–	**POOR**
22. F33 Grundkallan	0.928	–	–	0.362	**GOOD**
23. Skutskärsfjärden	0.546	0.253	0.800	–	**BAD**
24. K500 stations	0.515	0.373	–	–	**BAD**
25. Vallviksfjärden	0.879	0.556	–	0.300	**MODERATE**
26. Sandarnesfjärden	0.656	0.517	–	0.193	**POOR**
27. Skärsåfjärden	0.861	1.000	–	0.350	**GOOD**
28. N M Bottenhavets	0.786	–	–	0.469	**GOOD**
29. *Sundsvall Bay*	0.888	–	–	0.315	**GOOD**
30. Höga kusten (incl. C3)	0.864	0.818	–	0.354	**GOOD**
31. *Gavik*	0.905	–	–	0.377	**GOOD**
32. *Ömnefjärden*	0.667	–	–	0.415	**GOOD**
33. Örefjärden	0.832	0.571	–	0.215	**MODERATE**
🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	☐ No or insufficient data

Annex 1.4: **The Archipelago and Åland Seas (6)**

	Ecological Quality Ratio				**Overall eutrophication status**
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
34. *Archipelago Sea, outer, Kihti*	0.604	0.627	–	–	**MODERATE**
35. *Archipelago Sea, outer, Nötö*	0.585	0.427	–	–	**POOR**
36. *Archipelago Sea, inner, Turm Tryholm*	0.543	0.581	–	0.717	**MODERATE**
37. *Archipelago Sea, inner, Pala Tryholm*	0.269	0.380	–	–	**BAD**
38. Edeboviken & Granskär	0.429	–	–	0.129	**BAD**
39. *Archipelago Sea, inner, Hala*	0.271	0.440	–	–	**BAD**

🟦 High ecological status
🟩 Good ecological status
🟨 Moderate ecological status
🟧 Poor ecological status
🟥 Bad ecological status
☐ No or insufficient data

Annex 1.5: **Baltic Proper, northern parts (41)**

	Ecological Quality Ratio				**Overall eutrophication status**
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
40. NORTHERN BALTIC PROPER	0.523	0.231	–	0.000	**BAD**
41. W Torsbyholmen	0.522	0.168	–	0.270	**BAD**
42. *Ålvik*	0.367	0.134	–	–	**BAD**
43. Solöfjärden	0.422	0.188	–	0.190	**BAD**
44. Slussen	0.348	0.181	–	–	**BAD**
45. *Oxdjupet*	0.424	0.202	–	–	**BAD**
46. *Norra Vaxholmsfjärden*	0.474	0.195	–	–	**BAD**
47. *Lännerstasundet*	0.349	0.148	–	–	**BAD**
48. Koviksudde veckostation	0.458	0.158	–	–	**BAD**
49. *Koviksudde*	0.375	0.119	–	–	**BAD**
50. Karantänbojen	0.436	0.127	–	0.100	**BAD**
51. *Hammarby sjö*	0.348	0.122	–	–	**BAD**
52. *Halvkakssundet*	0.363	0.143	–	–	**BAD**
53. *Ekhagen*	0.322	0.099	–	–	**BAD**
54. *Brunnsviken Ekhagen*	0.442	0.164	–	–	**BAD**
55. *Blomskär*	0.480	0.148	–	–	**BAD**
56. *Blockhusudden*	0.349	0.151	–	–	**BAD**
57. Askrikefjärden	0.505	0.161	–	0.210	**BAD**
58. *Södra Vaxholmsfjärden*	0.517	0.178	–	–	**BAD**
59. Trällhavet II	0.461	0.258	–	0.214	**BAD**
60. *Trällhavet veckostation*	0.502	0.278	–	–	**BAD**
61. *Växlet, Fiskare*	0.332	1.000	–	–	**BAD**
62. *Växlet veckostation*	0.669	0.451	–	–	**POOR**
63. Ägnöfjärden	0.585	0.474	–	0.257	**POOR**
64. Åkerviksudde	0.690	0.556	–	–	**MODERATE**
65. Baggenfjärden	0.494	0.364	–	0.136	**BAD**
66. *Erstaviken*	0.645	0.543	–	–	**MODERATE**
67. *Farstaviken*	0.408	0.498	–	–	**BAD**
68. *Franska Stenarna*	0.675	0.488	–	–	**POOR**
69. *Ikorn*	0.623	0.519	–	–	**POOR**
70. *Kanholmsfjärden*	0.650	0.500	–	–	**POOR**
71. *NE Stora Möja*	0.888	0.556	–	–	**MODERATE**
72. *NW Eknö*	0.697	0.462	–	–	**POOR**
73. *Nyvarp*	0.512	0.272	–	–	**BAD**
74. *SE Österskär*	0.588	0.273	–	–	**BAD**
75. *Sollenkroka*	0.535	0.305	–	–	**BAD**
76. *GB 16 Bråvik L*	0.496	0.269	–	–	**BAD**
77. *GB 11 Bråvik Ö*	0.401	0.200	–	–	**BAD**
78. *GB 20 Bråvik P*	0.440	0.217	–	–	**BAD**
79. B1 and Knabbfjärden	0.564	0.648	0.920	0.379	**POOR**
80. Himmerfjärden	0.425	0.364	–	0.236	**BAD**

🟦 High ecological status
🟩 Good ecological status
🟨 Moderate ecological status
🟧 Poor ecological status
🟥 Bad ecological status
☐ No or insufficient data

Annex 1.6: **Gulf of Finland (20)**

	Ecological Quality Ratio				**Overall eutrophication status**
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
81. GULF OF FINLAND	0.468	0.220	–	0.394	**BAD**
82. Narva Bay	0.651	0.740	0.816	0.650	**MODERATE**
83. Tallinn Bay	0.648	0.455	0.715	0.460	**POOR**
84. *Western GoF, outer, Bågaskär*	0.445	0.190	–	0.162	**BAD**
85. *Western GoF, outer, Längden*	0.592	0.451	–	0.607	**POOR**
86. *Western GoF, inner, Tvärminne Storfjärd*	0.586	0.667	–	0.593	**MODERATE**
87. *Western GoF, inner, Pojo Bay*	0.573	0.394	–	0.599	**POOR**
88. *Gulf of Finland, outer, Länsi-Tonttu*	0.669	0.527	–	–	**MODERATE**
89. *Gulf of Finland, outer, Kirkonmaa*	0.522	0.543	–	–	**POOR**
90. *Gulf of Finland, outer, Huovari*	0.542	0.302	–	–	**BAD**
91. *Gulf of Finland, outer, Haapasaari*	0.585	0.416	–	0.230	**POOR**
92. *Gulf of Finland, outer Knapperskär*	0.604	0.411	–	–	**POOR**
93. *Gulf of Finland, inner, Porvoo*	0.558	0.326	–	0.017	**BAD**
94. *Gulf of Finland, inner, Vironlahti*	0.521	0.194	–	–	**BAD**
95. *Gulf of Finland, inner, Pyötsaari*	0.649	0.621	–	0.350	**MODERATE**
96. *Gulf of Finland, inner Ahvenkoski Bay*	0.523	0.261	–	0.435	**BAD**
97. *Gulf of Finland, inner Melkinselkä*	0.659	0.303	–	–	**BAD**
98. *Gulf of Finland, inner, Laajalahti*	0.253	0.126	–	–	**BAD**
99. Neva Bay	0.546	–	0.160	0.360	**BAD**
100. *Gulf of Finland, inner, Vasikkasaari*	0.446	0.401	–	–	**BAD**

🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	⬜ No or insufficient data

Annex 1.7: **Baltic Proper, Eastern Gotland Basin (9)**

	Ecological Quality Ratio				**Overall eutrophication status**
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
101. EASTERN GOTLAND BASIN	0.610	0.486	–	0.153	**BAD**
102. *SE GOTLAND BASIN, OPEN PARTS*	0.745	0.400	–	–	**POOR**
103. Open Baltic Sea northern coast	0.478	0.352	0.716	0.592	**BAD**
104. Open Baltic Sea southern coast	0.478	0.145	–	0.593	**BAD**
105. Plume from the Curonian Lagoon	0.499	0.162	0.824	0.840	**BAD**
106. Northern part of the Curonian Lagoon	0.403	0.031	0.300	0.450	**BAD**
107. Central part of the Curonian Lagoon	0.320	0.054	0.267	0.389	**BAD**
108. Kaliningrad region, open waters	0.129	0.158	–	–	**BAD**
109. Lithuanian open waters	0.470	0.382	–	–	**BAD**

🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	⬜ No or insufficient data

Annex 1.8: **Gulf of Riga (6)**

	Ecological Quality Ratio				**Overall eutrophication status**
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
110. GULF OF RIGA	0.543	0.340	–	–	**BAD**
111. Gulf of Riga, northern parts	0.651	0.540	0.603	0.550	**POOR**
112. Gulf of Riga, southern transitional waters	0.420	0.421	–	0.667	**POOR**
113. Väike Väin	0.764	0.708	0.519	0.600	**BAD**
114. Haapsalu Bay	0.409	0.221	0.511	0.560	**BAD**
115. Pärnu Bay	0.607	0.714	0.576	0.540	**POOR**

🟦 High ecological status
🟩 Good ecological status
🟨 Moderate ecological status
🟧 Poor ecological status
🟥 Bad ecological status
⬜ No or insufficient data

Annex 1.9: **Western Gotland Basin (20)**

	Ecological Quality Ratio				**Overall eutrophication status**
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
116. WESTERN GOTLAND BASIN	0.660	0.432	–	–	**POOR**
117. *Kristianopel KL 8*	0.274	0.098	–	–	**BAD**
118. S Kalmar	0.608	0.597	–	0.286	**POOR**
119. NW S Kalmar	0.479	0.443	–	0.336	**BAD**
120. Central Kalmar Sound	0.521	0.281	–	–	**POOR**
121. Northern Kalmar Sound	0.571	0.502	–	0.214	**POOR**
122. Misterhult	0.416	0.337	0.920	–	**BAD**
123. V 2	0.367	0.317	–	–	**BAD**
124. Lofta	0.530	0.463	–	–	**POOR**
125. *VA 11*	0.497	0.159	–	–	**BAD**
126. VA 04	0.432	0.167	–	0.392	**BAD**
127. *SÖ 13*	0.408	0.270	–	–	**BAD**
128. *VA 08*	0.632	0.178	–	–	**BAD**
129. *VA 10*	0.645	0.502	–	–	**POOR**
130. *SÖ 15*	0.524	0.293	–	–	**BAD**
131. *NO 03 Rimmö*	0.531	0.236	–	–	**BAD**
132. *VA 06*	0.652	0.240	–	–	**BAD**
133. *SÖ 14*	0.605	0.357	–	–	**BAD**
134. NO 01 Arkö	0.520	0.208	–	0.357	**BAD**
135. *VA 09*	0.692	0.223	–	–	**BAD**

🟦 High ecological status
🟩 Good ecological status
🟨 Moderate ecological status
🟧 Poor ecological status
🟥 Bad ecological status
⬜ No or insufficient data

Annex 1.10: Gulf of Gdansk (5)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
136. Gulf of Gdansk, Outer Puck Bay	0.745	0.348	–	0.878	BAD
137. Gulf of Gdansk, internal parts	0.859	0.431	–	–	POOR
138. Gulf of Gdansk, Vistula Profile	0.625	0.353	–	–	BAD
139. *Gdansk Deep*	0.782	0.452	–	0.000	BAD
140. Vistula Lagoon	0.577	0.143	–	–	BAD
🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	☐ No or insufficient data

Annex 1.11: Bornholm Basin (14)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
141. BORNHOLM BASIN	0.602	0.553	–	0.239	BAD
142. Polish coast. central parts	0.703	0.486	–	0.594	POOR
143. Pomeranian Bay, open part	0.760	0.465	–	0.640	POOR
144. Pomeranian Bay, Oder profile	0.524	0.399	–	–	POOR
145. Szczecin Lagoon	0.597	0.109	–	–	BAD
146. Pomeranian Bight	0.304	0.132	–	–	BAD
147. Western Hanö	0.623	1.000	–	0.236	MODERATE
148. Sölvesborg	0.489	1.000	–	–	BAD
149. Pukavik	0.529	0.902	–	0.429	POOR
150. Rönneby	0.514	0.453	–	–	POOR
151. *NY NV Aspö*	0.509	0.390	–	–	POOR
152. Outer Redden	0.583	0.397	–	0.329	POOR
153. Östra Fjärden	0.506	0.381	–	0.336	BAD
154. *Torhamns skärgård*	0.474	0.415	–	–	BAD
🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	☐ No or insufficient data

Annex 1.12: Arkona Basin (4)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
155. ARKONA BASIN	0.616	0.535	–	0.764	MODERATE
156. Drass-Zingst outer coast	0.573	0.848	–	–	MODERATE
157. Hjelm Bugt	0.533	0.838	0.702	–	POOR
158. Fakse Bugt - Stevns	0.336	0.843	0.809	–	BAD
🟦 High ecological status	🟩 Good ecological status	🟨 Moderate ecological status	🟧 Poor ecological status	🟥 Bad ecological status	☐ No or insufficient data

Annex 1.13: Kiel Bight and Mecklenburg Bight (5)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
159. Wismar Bight	0.359	0.366	–	–	BAD
160. Lübeck Bight	0.672	0.897	0.489	–	BAD
161. Mecklenburg Bight	0.461	0.573	–	–	POOR
162. Fehmarn Belt	0.621	0.778	0.543	–	POOR
163. Kiel Bight, SW parts	0.646	0.786	0.447	–	BAD

🟦 High ecological status 🟩 Good ecological status 🟨 Moderate ecological status 🟧 Poor ecological status 🟥 Bad ecological status ⬜ No or insufficient data

Annex 1.14: Danish Straits including the Sound (10)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
164. GREAT BELT	0.356	0.295	–	–	BAD
165. NW Kiel Bight and Flensburg Fjord, o.p.	0.611	0.648	0.388	–	BAD
166. The Sound, central parts	0.584	0.523	0.779	–	POOR
167. Lillebælt, southern parts	0.456	0.268	0.581	–	BAD
168. Odense fjord	0.282	0.464	0.366	–	BAD
169. North of Fyn	0.537	0.349	0.579	0.577	BAD
170. Aarhus Bay	0.527	0.602	0.663	–	POOR
171. Northern Sound	0.748	0.593	–	–	MODERATE
172. Central Sound	0.542	0.557	–	–	POOR
173. Southern Sound	0.559	0.930	–	–	POOR

🟦 High ecological status 🟩 Good ecological status 🟨 Moderate ecological status 🟧 Poor ecological status 🟥 Bad ecological status ⬜ No or insufficient data

Annex 1.15: The Kattegat (16)

	Ecological Quality Ratio				Overall eutrophication status
	Primary signals		Secondary signals		
	PC	PP	SAV	BIC	
174. KATTEGAT, NORTHERN WESTERN	0.845	0.603	–	–	MODERATE
175. KATTEGAT NORTH EASTERN	0.787	0.813	–	–	GOOD
176. KATTEGAT, CENTRAL	0.697	0.440	–	0.549	POOR
177. KATTEGAT, SOUTH EASTERN	0.821	0.588	–	–	MODERATE
178. KATTEGAT, SOUTH OPEN PARTS	0.561	0.351	–	–	BAD
179. KATTEGAT, SOUTH WESTERN	0.716	0.460	–	0.584	BAD
180. Kattegat. southern coastal parts	0.540	0.398	–	–	POOR
181. Roskilde Fjord	0.763	–	0.644	0.412	BAD
182. Isefjorden	–	–	0.894	0.029	BAD
183. Mariager fjord	0.519	–	–	0.369	BAD
184. Randers fjord	0.369	0.562	0.311	0.578	BAD
185. Limfjorden	0.650	–	0.420	0.326	BAD
186. Kattegat, western coastal parts	0.739	0.570	–	–	MODERATE
187. Kattegat, north east inshore	0.709	0.508	–	–	MODERATE
188. *Laholm Bight*	0.769	0.592	–	–	MODERATE
189. Skälderviken	0.647	0.369	–	–	BAD

🟦 High ecological status 🟩 Good ecological status 🟨 Moderate ecological status 🟧 Poor ecological status 🟥 Bad ecological status ⬜ No or insufficient data

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