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Approaches and methods for eutrophication target setting in the Baltic Sea region

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NOTE: It should be noted that due to an oversight, data from the German territorial waters and EEZ were not sufficiently included in this study. Therefore, the targets presented in this report and their derivations are considered preliminary by Germany. Recalculations including these data were carried out and they are included in document 3/9, Add.1 of the HELCOM CORE EUTRO 7/2012 workshop; however, due to time constraints they could not be included in this report.

Preface

This report is the result of the project "Review of the ecological targets for eutrophication of the HELCOM BSAP", abbreviated to HELCOM TARGREV. The objectives have been to revise the scientific basis underlying the ecological targets for eutrophication, placing much emphasis on providing a strengthened data and information basis for the setting of quantitative targets. The results are first of all likely to form the information basis on which decisions with regard to reviewing and, if necessary, revising the maximum allowable inputs of nutrients in the Baltic Sea Action Plan, including the provisional country-wise nutrient reduction figures, will be made. In addition, the results quantitatively define HELCOM's ecological targets for eutrophication and the indicators can be used for assessment of the eutrophication status of the Baltic Sea. Hence, HELCOM TARGREV is an important project since the results should ultimately ensure an appropriate set of measures to improve the eutrophication status of the Baltic Sea.

The Baltic Sea Action Plan, adopted at the HELCOM Ministerial Meeting in Krakow, Poland in 2007 (HELCOM 2007a), has the following overarching vision for the Baltic Sea:

A healthy Baltic Sea environment with diverse biological components functioning in balance, resulting in a good ecological status and supporting a wide range of sustainable human economic and social activities.

The Baltic Sea Action Plan (BSAP) implements the Ecosystem Approach (EA) to the management of human activities affecting the health of the Baltic Sea. The Action Plan focuses on four thematic issues (also referred to as segments): eutrophication, hazardous substances, maritime activities and biodiversity. The eutrophication segment is hierarchal, with the strategic goal for eutrophication being "*The Baltic Sea unaffected by eutrophication*". This goal is subsequently being defined by five ecological objectives: (1) Concentration of nutrients close to natural levels; (2) Clear water; (3) Natural level of algal blooms; (4) Natural distribution and occurrence of plants and animals; and (5) Natural oxygen levels.

Implementing the BSAP and the EA would ideally include the following activities: (1) Agreeing on principles for target setting in regard to nutrients, clear water, algae, submerged aquatic vegetation, benthic invertebrates and oxygen; (2) Estimations of critical loads (threshold values) per basin and per objective; and (3) Overlay of the critical loads per basins in order to estimate the load reductions needed to fulfil all ecological targets. In practice, the objective most sensitive to nutrient inputs will be decisive for the calculation of the load reductions required.

The HELCOM BSAP is based on just one of the five ecological objectives, "clear water", which in practice is equivalent to "light penetration", measured as Secchi depth. As this target has been considered preliminary, the subsequent estimation of critical loads (total allowable loads) as well as the country-wise allocation of the critical loads also has to be regarded as preliminary.

At the time of the adoption of the BSAP, it was recognised that additional actions were required to review and strengthen the basis for calculating maximum allowable inputs and country-wise load allocations. Baltic Sea countries have by initiating HELCOM TARGREV established a process which, as a first step, will establish a science-based foundation for the calculation of total allowable loads and their country-wise allocation.

Executive Summary

This report describes the outcome of the project "Review of the ecological targets for eutrophication of the HELCOM BSAP", also known as HELCOM TARGREV. The objectives of HELCOM TARGREV have been to revise the scientific basis underlying the ecological targets for eutrophication, placing much emphasis on providing a strengthened data and information basis for the setting of quantitative targets. The results are first of all likely to form the information basis on which decisions in regard to reviewing and if necessary revising the maximum allowable inputs (MAI) of nutrient of the Baltic Sea Action Plan, including the provisional country-wise allocation reduction targets (CART), will be made.

Background

Nutrient enrichment and the abatement of eutrophication effects has been an issue for decades in the Baltic Sea region. Significant efforts and resources have been spent on research, monitoring and assessment as well as on the reduction of losses, discharges and emissions of nitrogen and phosphorus. Our understanding of the links between human activities causing eutrophication and the structures and functions of Baltic marine ecosystems is well developed compared to most other marine regions.

HELCOM has recently produced a comprehensive and integrated thematic assessment of the effects of nutrient enrichment in the Baltic Sea region. The eutrophication status has been assessed and classified in 189 "areas" of the Baltic Sea, of which 17 are open and 172 are coastal areas. The open waters in the Bothnian Bay and in the Swedish parts of the north-eastern 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 north-eastern Kattegat is influenced by Atlantic waters. Open waters of all other basins are classified as "areas affected by eutrophication".

Once an area is identified as being "affected by eutrophication", the Baltic Sea states are required to implement measures to abate eutrophication, e.g. via the Baltic Sea Action Plan, HELCOM Recommendations or in the case of those countries also being EU Member States, via implementation of relevant Directives. The Baltic Sea Action Plan (BSAP), which implements the Ecosystem Approach (EA) to management of human activities affecting the health of the Baltic Sea, focuses on four thematic issues (also referred to as segments), e.g. eutrophication, hazardous substances, maritime activities and biodiversity. The eutrophication segment is hierarchal, with the strategic goal for eutrophication being "*The Baltic Sea unaffected by eutrophication*", which is subsequently being defined by five ecological objectives. The Directives concerning eutrophication are: (1) The EC Urban Waste Water Treatment Directive; (2) the EC Nitrates Directive; (3) the EU Water Framework Directive; and (4) the EU Marine Strategy Framework Directive.

The above introduced policies all relate to nutrient enrichment and eutrophication, and include goals and targets concerning the eutrophication status of marine waters. It is widely accepted that the goals and targets converge in practice.

Temporal trends and identification of thresholds

Nutrient inputs to the Baltic Sea have increased multifold over the 20th century and this affected nutrients, phytoplankton, oxygen, water transparency and benthic invertebrates. The analyses of long-term trends described in this report identify three distinct periods: (1) a pre-eutrophication period before ca. 1940; (2) a eutrophication period from ca. 1940 to ca. 1980; and (3) a so-called eutrophication stagnation period from ca. 1980 to present, bearing in mind that eutrophication is an increase in the organic input to the Baltic Sea. It should also be acknowledged that the Baltic Sea was affected by human activities in the pre-eutrophication period, although to a much smaller extent than at present. The intention of the BSAP is to initiate an oligotrophication period, i.e. a period characterised by a reduction in the allochthonous and autochthonous organic input to the Baltic Sea.

Secchi depths, representing the target "clear water", have declined significantly in all sub-basins of the Baltic Sea over the last 100 years, mostly in response to eutrophication but possibly also due to increased inputs of coloured dissolved organic material from land, most pronounced in the Bothnian Bay and the Gulf of Finland. In these two sub-basins, it is estimated that this change could account for an almost 0.5 m decline in Secchi depth. Oxygen concentrations in the bottom waters of the Baltic Sea have deteriorated enormously, and a large oxygen debt, proposed as a new indicator, has accumulated over the last 100 years, particularly in the Bornholm Basin and the Baltic Proper. Nutrient and Chlorophyll *a* data are available from around 1970 onwards and can be used to describe the later part of the eutrophication period and the eutrophication stagnation period only. Species diversity of benthic invertebrates has decreased in certain sub-basins in response to deteriorating oxygen conditions.

Three state-of-the-art biogeochemical models for the Baltic Sea have been used to simulate the current status of various eutrophication indicators as well as the status believed to be present around 1900. This ensemble modelling approach yielded consistent estimates for the inorganic nutrients, whereas Chlorophyll *a* and Secchi depths varied considerably across the models due to model differences.

Improved evidence for eutrophication target setting

The indicator distributions during the pre-eutrophication period has been used for suggesting targets using the criterion that exceeding the 95% confidence interval of the 'natural' variation during this period would signify a significant deviation from a relatively unaffected situation. This approach was successfully applied to the Secchi depth and oxygen debt, and the suggested targets derived are considered well-founded and recommended as absolute targets. A simpler approach was employed for nutrients and Chlorophyll *a* by averaging the ensemble model predictions characterising the levels around 1900 with the estimated indicator levels from the 1970s. These targets are not as scientifically well-founded as those for Secchi depth and oxygen debt, and therefore recommended as guiding targets. Consequently, targets have been suggested for four out of HELCOMs five ecological objectives, which are presented in the conclusion.

The analyses and the proposed targets have been developed for the basins used in the BALTSEM model, which will be used for calculating MAI and CART. However, for the purpose of assessing the state of eutrophication in the Baltic Sea, the proposed targets have also been recalculated for the HELCOM sub-divisions.

Comparing the suggested targets with the present observed and modelled status confirmed that all sub-basins of the Baltic Sea are affected to varying degrees by eutrophication. The comparison also indicated that there could be systematic biases between assessing the status using indicators and estimating the status by the BALTSEM model, which will be employed for the revision of the BSAP. It is recommended to further analyse these potential biases and establish an intercalibration between the targets based on indicators and models used for estimating maximum allowable inputs.

The revision of the ecological targets presented in this report is believed to provide sufficient basis for revising the estimated maximum allowable inputs to each of the sub-basins, and subsequently calculating country-specific nutrient reduction targets.

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1. Introduction

Nutrient enrichment and eutrophication has been an issue in the Baltic Sea region for decades. Significant efforts and resources have been spent on research, monitoring, assessment and reduction of losses, discharges and emissions of nitrogen and phosphorus.

Our conceptual understanding of the links between human activities causing eutrophication and the structures and functions of Baltic marine ecosystems is well developed compared to other marine regions. However, for management purposes the quantification of such links with low uncertainty and concrete quantitative objectives are still lacking. Hence, a key issue still to be addressed is the setting of evidence-based eutrophication targets.

Definitions used in the target setting approach are described in detail in Annex A.

1.1 Eutrophication in the Baltic Sea

Eutrophication signals and trends have been monitored and assessed by the countries surrounding the Baltic Sea for decades. There is a consensus among the Baltic Sea states that eutrophication is a large-scale problem and that all shoreline states must reduce inputs of nutrients.

HELCOM has recently produced a comprehensive and integrated thematic assessment of the effects of nutrient enrichment in the Baltic Sea region (HELCOM 2009, Andersen et al. 2011). The eutrophication status has been assessed and classified in 189 "areas" of the Baltic Sea, of which 17 are open and 172 are coastal areas.

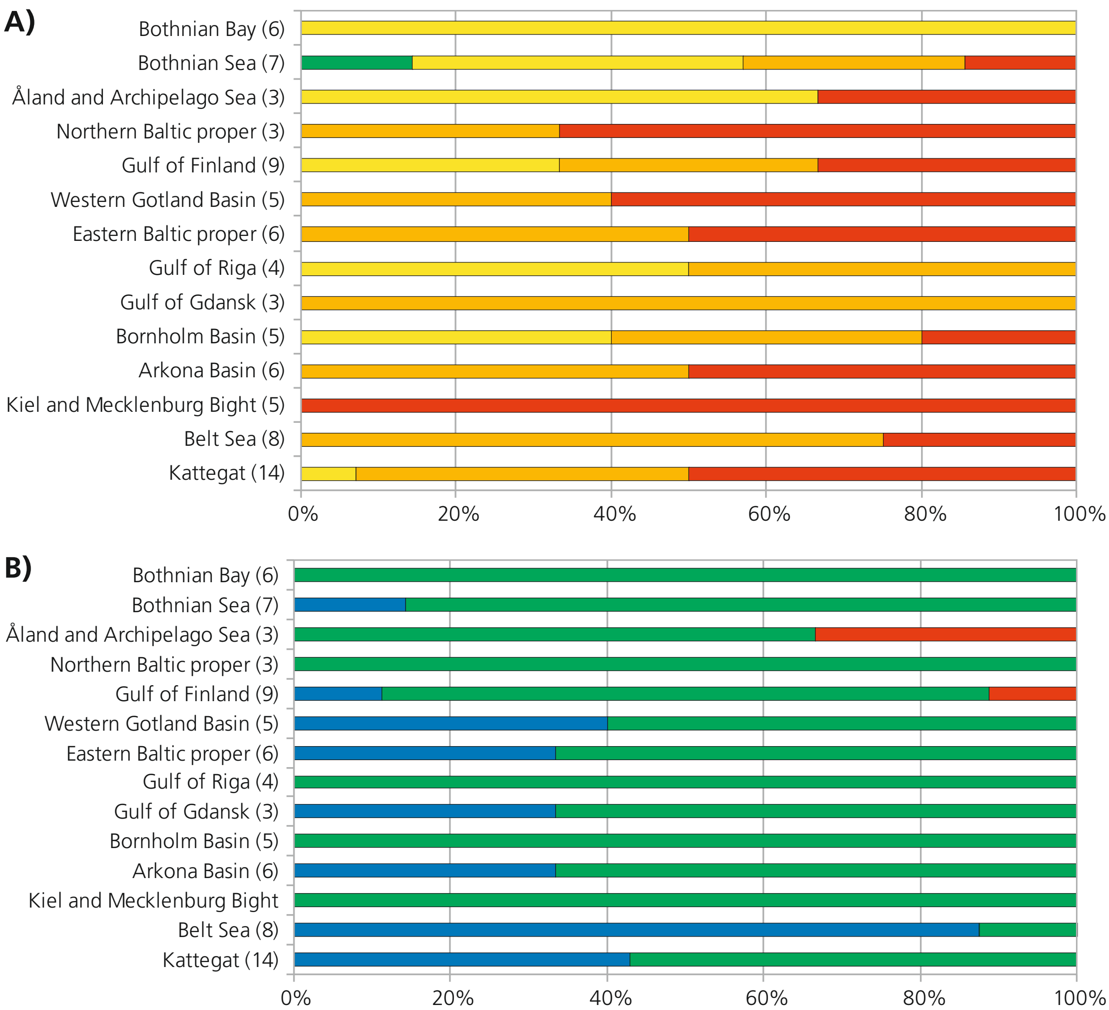


Figure 1.1 *Classification of eutrophication status in the basins of the Baltic Sea (Panel A) and estimation of the confidence of the classifications made (Panel B). From HELCOM (2010), based on HELCOM (2009), Andersen et al. (2011). Colours follow the WFD classification, i.e. blue=high; green=good; yellow=moderate; orange=poor; and red=bad.*

The open waters in the Bothnian Bay and in the Swedish parts of the north-eastern 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 north-eastern 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* (Chl *a*) concentrations. For coastal waters, eleven have been classified as “areas not affected by eutrophication” and 161 as “areas affected by eutrophication”. A summary of this assessment is presented in Fig. 1.1. The geographical variations in eutrophication status are shown in Fig. 1. 2.

The Baltic Sea has been sub-divided into 13 basins corresponding to the spatial resolution of the BALTSEM model, which will be used for revising the BSAP maximum allowable inputs and country-specific nutrient reductions required to achieve targets proposed in this report. Although this spatial sub-division does not exactly match HELCOM’s spatial sub-division, the main objective of TARGREV is to deliver targets that can be implicitly used in the revision of the BSAP. However, since the models developed in TARGREV contain a spatial component, it is possible to translate targets from the BALTSEM sub-division into another spatial division. TARGREV only addresses the open waters of the Baltic Sea (see Section 2.2).

The 13 basins in the BALTSEM model are numbered according to the following scheme, which will also be adopted in this report: 1=Northern Kattegat; 2= Central Kattegat; 3=Southern Kattegat; 4=Northern Belt Sea; 5=Southern Belt Sea; 6=The Sound; 7=Arkona Basin; 8=Bornholm Basin; 9=Baltic Proper; 10=Bothnian Sea; 11=Bothnian Bay; 12=Gulf of Riga; and 13=Gulf of Finland. For some analyses, these basins have been aggregated so that the Kattegat refers to basins 1-3; the Danish Straits refers to basins 4-6; and for the statistical analysis of oxygen (see Section 2.4) the Baltic Proper (basin 9) and Gulf of Finland (basin 13) have been aggregated.

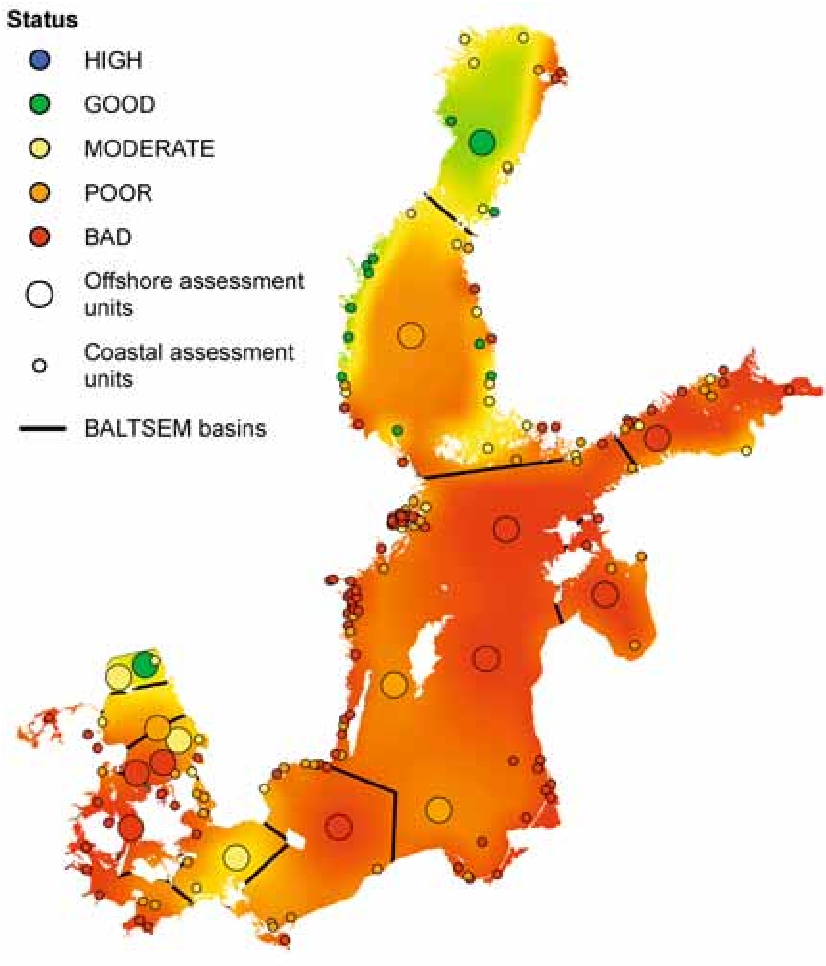


Figure 1. 2 *Classification of eutrophication status in the Baltic Sea and its subdivisions* sensu *the BALTSEM model, which has been used in this report and will be used for the calculation of total allowable loads and their country-wise allocation. Based on HELCOM (2010).*

1.2 Policy context

The target setting for HELCOM’s eutrophication objectives in the Baltic Sea, or parts hereof, is required by a suite of policies such as the Baltic Sea Action Plan (HELCOM 2007a), the EU Water Framework Directive (Anon. 2000) and the EU Marine Strategy Framework Directive (Anon. 2008).

1.2.1 The HELCOM Baltic Sea Action Plan

The HELCOM Baltic Sea Action Plan (BSAP) is an ambitious strategy outlining visions, goals and objectives to restore good ecological status of the Baltic marine environment by 2021.

The BSAP has an overarching vision of "*a healthy Baltic Sea, with diverse biological components functioning in balance, resulting in a good ecological status and supporting a wide range of sustainable human, economic and social activities*" (HELCOM 2007a).

The eutrophication segment, which is of interest in the context of the HELCOM TARGREV project, is hierarchical with the strategic goal for eutrophication being "The Baltic Sea unaffected by eutrophication". The goal is subsequently defined by five ecological objectives (see Introduction). The currently used target values for "Clear water", on which the calculation of maximum allowable loads of the BSAP is mainly based, are modelled values but they have been validated against those *in situ* values that originate from the HELCOM EUTRO project as presented in "Development of tools for assessment of eutrophication in the Baltic Sea", which was published in HELCOM (2006). The objective of HELCOM EUTRO was merely to develop and test a simple indicator-based tool enabling a harmonised Baltic Sea-wide assessment of eutrophication. One of the indicators used was Secchi depth, a proxy of "Clear water". Data about basin-specific Secchi depth reference conditions were collated and combined with other indicators to demonstrate and test what ultimately turned into the HELCOM Eutrophication Assessment Tool, abbreviated to HEAT. For Secchi depth, an acceptable deviation from reference conditions was tentatively set as a -25% deviation from reference conditions. More information about HELCOM EUTRO and the data used can be found in HELCOM (2006). An updated data set and a detailed description of the HEAT tool can be found in HELCOM (2009) and Andersen et al. (2011).

The BSAP contains measures that in 2007 were estimated to be sufficient to reduce eutrophication to a target level that would correspond to good ecological/environmental status by the year 2021 (HELCOM 2007a). It was estimated that nutrient load reductions of 135,000 tonnes for nitrogen and 15,250 tonnes for phosphorus would be needed relative to a baseline period (1997–2003). The largest reductions were on loads to the Baltic Proper, while the Gulf of Bothnia was during the preparation of the BSAP considered to be in good ecological/environmental status and thus not in need of nutrient reductions. It was estimated that the reductions would result in achieving the eutrophication-related targets on water transparency (Wulff et al. 2007). However, this assumption was questioned by HELCOM (2009), where the open parts of the Bothnian Sea were classified as affected by eutrophication (cf. Fig. 1.2).

Table 1.1 summarizes the inputs to and outputs from the MARE/NEST calculations on maximum allowable inputs to achieve "good environmental status" while Table 1.2 indicates the provisional nutrient reduction requirements of the countries that are based on the maximum allowable nutrient inputs in Table 1.1.

Table 1.1 *Provisional maximum allowable inputs of phosphorus and nitrogen to achieve "good ecological status" (calculated for water transparency) and corresponding minimum load reductions (in tonnes) calculated per sub-basin as agreed in the BSAP (HELCOM 2010).*

	Maximum allowable nutrient loads (tonnes)		Inputs in 1997–2003		Needed reductions	
	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

Table 1.2 *Provisional country-wise nutrient load reduction allocations, in tonnes (HELCOM 2007a).*

	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

It should be emphasised that updated calculations of maximum allowable inputs and their country-wise allocation are not a part of HELCOM TARGREV. However, the revision of the ecological targets will provide the necessary information to revise the estimated maximum allowable inputs to each of the sub-basins and subsequently calculating country-specific nutrient reduction targets. The calculations of maximum allowable inputs and country-specific load reduction targets will be made by the Baltic Nest Institute at Stockholm University, Sweden.

1.2.2 Eutrophication-related EU Directives

The EU Marine Strategy Framework Directive (MSFD), or in full "Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy" (Marine Strategy Framework Directive), entered into force on 15 July 2008 (Anon. 2008).

The MSFD Directive focuses on implementing an ecosystem-based approach to the management of the human activities and pressures affecting the marine environment.

In principle, the MSFD covers all European marine waters including coastal waters (the later only in regard to issues not dealt with by the Water Framework Directive) and has as an overarching aim of reaching or maintaining "good environmental status" in all European marine waters by 2020.

As a preparatory action to the above, the MSFD required that the European Commission by 15 July 2010 should lay down both criteria and methodological standards to allow consistency in the approach, by which EU Member States (MS) assess the extent to which Good Environmental Status (GES) is being achieved. Scientific advice for guidance on this was sought from expert groups coordinated by the International Council for the Exploration of the Sea (ICES) and the EU's Joint Research Centre (JRC) to provide scientific support for the European Commission in meeting this obligation. A Eutrophication Task Group dealing with Descriptor 5 - "eutrophication" - was established as well as task groups for most of the other MSFD descriptors.

Currently, the following two reports can support the process of setting eutrophication targets for the open parts of the Baltic Sea: 1) Scientific support to the European Commission on the Marine Strategy Framework Directive. Management Group Report (EU and ICES 2010); and 2) Task Group 5 Report – Eutrophication - JRC European Commission and ICES (Ferreira et al. 2010, summarized by Ferriera et al. 2011).

The European Commission, based on the above reports, adopted a decision on the criteria of good environmental status in marine waters (Anon. 2010), which in regard to "Descriptor 5: Human-induced eutrophication" reads:

"The assessment of eutrophication in marine waters needs to take into account the assessment for coastal and transitional waters under Directive 2000/60/EC (Annex V, 1.2.3 and 1.2.4) and related guidance, in a way which ensures comparability, taking also into consideration the information and knowledge gathered and approaches developed in the framework of regional sea conventions. Based on a screening procedure as part of the initial assessment, risk-based considerations may be taken into account to assess eutrophication in an efficient manner. The assessment needs to combine information on nutrient levels and on a range of those primary effects and of secondary effects which are ecologically relevant, taking into account relevant temporal scales. Considering that the concentration of nutrients is related to nutrient loads from rivers in the catchment area, cooperation with landlocked Member States using established cooperation structures in accordance with

the third subparagraph of Article 6(2) of Directive 2008/56/EC is particularly relevant.

5.1. Nutrients levels

- *Nutrients concentration in the water column (5.1.1)*
- *Nutrient ratios (silica, nitrogen and phosphorus), where appropriate (5.1.2)*

5.2. Direct effects of nutrient enrichment

- *Chlorophyll concentration in the water column (5.2.1)*
- *Water transparency related to increase in suspended algae, where relevant (5.2.2)*
- *Abundance of opportunistic macroalgae (5.2.3)*
- *Species shift in floristic composition such as diatom to flagellate ratio, benthic to pelagic shifts, as well as bloom events of nuisance/toxic algal blooms (e.g. cyanobacteria) caused by human activities (5.2.4)*

5.3. Indirect effects of nutrient enrichment

- *Abundance of perennial seaweeds and seagrasses (e.g. fucoids, eelgrass and Neptune grass) adversely impacted by decrease in water transparency (5.3.1)*
- *Dissolved oxygen, i.e. changes due to increased organic matter decomposition and size of the area concerned (5.3.2).*"

Hence, the state of play in regard to the MSFD is currently that the Commission has decided on criteria on a general level, supplemented by a total of three eutrophication criteria each with a set of sub-criteria. The Commission Decision describes neither methodological standards nor detailed standards for the definition of "Good Environmental Status" in regard to eutrophication, instead the general guidance given in the decision is to be implemented by the Member States consistently across marine regions.

The EU Water Framework Directive (WFD), in full "Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy", was adopted by the European Parliament and the EU Council in 2000 (Anon. 2000). The WFD covers groundwater, inland waters (rivers and lakes), transitional waters (estuaries) and coastal marine waters.

An overarching aim of the WFD is that all European waters should be classified as having "good ecological status" by the end of 2015. The ecological targets of the WFD are indirectly defined for a number of biological quality elements (phytoplankton, macroalgae and angiosperms, benthic invertebrate fauna, and fish, the later only applicable for transitional waters) by so-called "normative definitions" (Table 1.3).

Table 1.3 *The "normative definitions" for the coastal biological quality elements in the WFD.*

Phytoplankton	Macroalgae and angiosperms	Benthic invertebrate fauna
The composition and abundance of phytoplanktonic taxa: 1. *are consistent with undisturbed conditions; or* 2. *show slight signs of disturbance; or* 3. *show signs of moderate disturbance.* Cases 1 and 2 above represent high and good ecological status, respectively and are considered as fulfilment of the targets. Case 3 represents moderate ecological status, which is equivalent to impaired conditions. The average phytoplankton biomass: 1. *is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions;* 2. *there are slight changes in biomass compared to type-specific conditions; such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the quality of the water;* 3. *the algal biomass is substantially outside the range associated with type-specific conditions, and is such as to impact upon other biological quality elements.* Case 1 and 2 represent high and good ecological status, respectively. Case 3 represents moderate ecological status, which is equivalent to impaired conditions. Planktonic blooms 1. *occur at a frequency and intensity which is consistent with the type specific physico-chemical conditions;* 2. *a slight; or* 3. *moderate increase in the frequency and intensity of the type-specific planktonic blooms may occur;* 4. *persistent blooms may occur during summer months.* Cases 1 and 2 represent high and good ecological status, respectively. Cases 3 and 4 represent moderate ecological status.	All disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions: 1. *are present;* 2. *most disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present;* 3. *a moderate number of the disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are absent.* Cases 1 and 2 represent high and good ecological status, respectively. Case 3 represents moderate ecological status. The level of macroalgal cover and angiosperm abundance: 1. *is consistent with "undisturbed conditions"; or* 2. *shows slight signs of disturbance;* 3. *the macroalgal cover and angiosperm abundance is moderately disturbed and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.* Cases 1 and 2 represent high and good ecological status, respectively. Case 3 represents moderate ecological status.	The level of diversity and abundance of invertebrate taxa is: 1. *within; or* 2. *slightly outside; or* 3. *moderately outside the range normally associated with undisturbed conditions.* Cases 1 and 2 represent high and good ecological status, respectively. Case 3 represents moderate ecological status. In regard to the disturbance-sensitive taxa associated with undisturbed conditions: 1. *all; or* 2. *most of the taxa are present;* 3. *taxa indicative of pollution are present and many of the sensitive taxa of the type-specific communities are absent.* Cases 1 and 2 represent high and good ecological status, respectively. Case 3 represents moderate ecological status.

The implementation of the WFD – including the target setting, in a WFD context named 'boundary setting' – has been coordinated and harmonised via a Common Implementation Strategy (CIS) since 2000. This CIS process has resulted in a variety of reports, including descriptions of how the directive should be interpreted and implemented.

The WFD guidance is useful for setting evidence-based Baltic Sea-specific targets in regard to eutrophication. Much of HELCOM's ongoing work is already directly or indirectly linked to Member States' implementation of the WFD. For example, it was specified that HELCOM's integrated thematic assessment of eutrophication in the Baltic Sea region should take into account both the Baltic Sea Action Plan (for both open and coastal waters) and the WFD (for coastal waters). Hence, the target setting principles used for the open parts of the Baltic Sea are, in principle, consistent with the principles used by the EU Member States implementing the WFD for coastal and transitional waters (HELCOM 2009).

A suite of other EU Directives besides the WFD is relevant in regard to the management of coastal eutrophication and target setting. These directives are briefly summarised below.

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

Directive 91/271/EEC of 21 May 1991 concerning urban waste water treatment (Anon. 1991b): The objective of the Urban Wastewater Directive 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 the assessment of the sensitivity of the receiving waters. The Member States shall identify areas which are sensitive in terms of eutrophication. Competent authorities shall monitor discharges and waters subject to discharges.

The above introduced policies all relate to nutrient enrichment and eutrophication and do include goals and targets in regard to eutrophication status of marine waters. It is widely accepted, e.g. HELCOM (2009), HELCOM (2010), that the goals and targets in practice converge as illustrated in Fig. 1.3.

DRIVER	STATUS CLASSIFICATION	
	Unaffected/Acceptable	Affected/Unacceptable
BSAP	Unaffected by eutrophication	Affected by eutrophication
MSFD	Good Environmental Status	Polluted
WFD	High and Good Ecological Status	Moderate, Poor, and Bad Ecological Status
UWWTD	Un-polluted/non-sensitive	Polluted/sensitive
ND	Un-polluted	Polluted

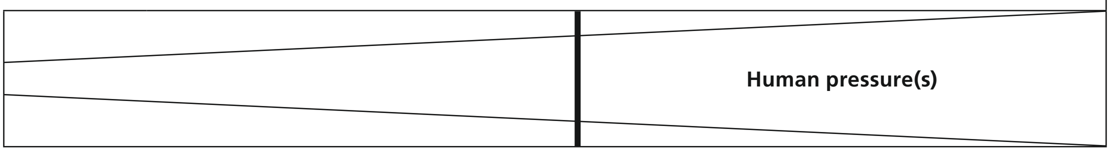


Figure 1.3 *Relationships between the HELCOM Baltic Sea Action Plan and the relevant European water policy directives with direct focus on eutrophication status. BSAP = Baltic Sea Action Plan; MSFD = Marine Strategy Framework Directive; WFD = Water Framework Directive; UWTTD = Urban Waste Water Treatment Directive; ND = Nitrates Directive; ES = Ecological Status* sensu *the Water Framework Directive. Based on HELCOM (2009).*

1.3 Towards evidence-based eutrophication targets for the open parts of the Baltic Sea

The simplest way to establish a target is to analyse all available data and to categorise them. Well-known examples include Nixon (1995) and a Baltic Sea-specific example by Wasmund et al. (2001). The derived assessment criteria, where the boundary between oligotrophic and mesotrophic can be regarded as the eutrophication targets, are summarised in Table 1.4. Please refer to the original publication for descriptions of the classifications.

A better justified approach is to analyse all available data and to base the categorisation or target setting on information of uncertainties as done in the case of benthic invertebrates in the Baltic Sea (HELCOM 2009, Vilnäs & Norkko 2011, see also Section 2.5 for details). Here, the historical data are regarded as "reference conditions" and the uncertainties as an "acceptable deviation" from the reference conditions (See Annex A for a definition of these concepts). The term "reference conditions" should by no means be interpreted as pristine conditions.

Currently, approaches to translate "reference conditions" and "acceptable deviations" into specific quantifiable targets are few and mostly heuristic, limited and mostly related to either the implementation of the Water Framework Directive or HELCOM's integrated thematic assessment of eutrophication status in the Baltic Sea.

The HELCOM Baltic Sea Action Plan specifies the goals but provides no guidance in regard to target setting. However, HELCOM (2006) can be used as an indirect guide with regard to defining good ecological status. Two elements are of particular interest. First, the step-wise approach where the vision, strategic goals and ecological objectives are included in the BSAP, whilst the selection of indicators and setting the targets are carried out separately (in the HELCOM CORESET project, in the HELCOM TARGREV project, and indirectly also in HELCOM's thematic assessments, e.g. in the HELCOM EUTRO-PRO project 2006-2009). Second, the approach of determining reference conditions and acceptable deviations, which are used by HELCOM's "integrated thematic assessment of eutrophication status" and summarised in Fig. 1.4.

Table 1.4 *Examples of predefined assessment criteria.*

	Organic Carbon Supply g C m^{-2} y^{-1}	Primary production g C m^{-2} y^{-1}	Chlorophyll *a* mg m^{-3}
	(Nixon 1995)	(Wasmund et al. 2001)	
Oligotrophic	< 100	< 100	< 0.8
Mesotrophic	100-300	100-250	> 0.8-4
Eutrophic	301-500	250-450	4-10
Poly/hyper-trophic	> 500	> 450	> 10

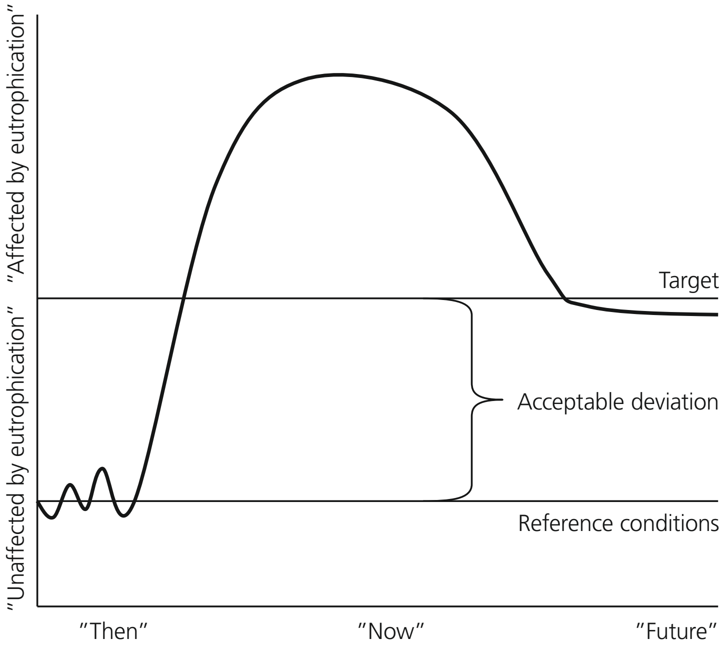


Figure 1.4 *Conceptual model of the current target setting concept, where the target is defiend as reference conditions (the "then" situation) ± an acceptable deviation (here the "now" situation being the prevailing conditions).*

The implication of HELCOM (2006) in combination with HELCOM (2007a) and HELCOM (2009) is a *de facto* acceptance of using the concepts and most importantly the combination of reference conditions and acceptable deviations for target setting. An added value is a harmonisation with the implementation process as well as the assessment principles of the WFD.

The concept of "acceptable deviation" has a number of strengths. It allows setting specific quantitative targets that enable the classification of the environmental/ecological status. It is also a widely used concept, e.g. by HELCOM (the integrated thematic assessment of eutrophication in the Baltic Sea region) and by the WFD for coastal and transitional waters. Over the last decade, the

amount of scientific literature on the understanding of "acceptable deviations" and target setting in coastal waters has increased significantly. It is also important to note that the "acceptable deviations" set for biological parameters in coastal and transitional water bodies and/or types have been or are being intercalibrated in the context of the WFD. Further, it should be emphasised that HELCOM's integrated thematic assessment of eutrophication, in particular the classification of eutrophication status (HELCOM 2009), is based on basin-, site- or water body-specific information on acceptable deviations.

Some weaknesses of the concept are identified. Although an increasing proportion of the values (%) for "acceptable deviation" are based on scientific analyses, not all values for "acceptable deviations" are scientifically based. Hence, the degree of expert judgement ought to be further reduced. There also seems to be a lack of understanding amongst (some) scientists that target setting is a multi-step process where the basis (being the initial steps) is scientific information, but the final setting (ultimate step) is a decision-making process converging the best available scientific information with what is practicably possible. Further, the current deficit of science in regard to setting "acceptable deviations" (and targets) dilutes the Ecosystem Approach to an extent where it has limited meaning.

The concepts of "reference conditions" and "acceptable deviations" are well defined and widely used, e.g. in HELCOM's thematic assessment of eutrophication status and by EU Member States in their implementation of the Water Framework Directive for coastal and transitional waters. Hence, the concepts should be used by HELCOM TARGREV as a first step for setting up normative definitions for each individual eutrophication objective of the HELCOM Baltic Sea Action Plan.

The suggested tentative normative definitions of targets (Table 1.5) should be seen as a first step towards defining the operational targets. HELCOM TARGREV's planned analysis of temporal trends for selected eutrophication indicators and the planned modelling will provide a scientific basis for setting operational basin-wise or sub-basin-wise targets.

The working hypothesis has been that the Baltic Sea ecosystem(s) can cope with (some) human activities and pressures, but only to a certain extent. Above a certain level of pressure, ecological effects become pronounced and the system collapses. In case there is a gradual response to nutrient inputs and nutrient enrichment, and thus no 'break points', targets will have to be based on the concept of reference conditions, perhaps being the early 1900s, and acceptable deviations.

Regarding the case of non-linearity and distinct ecosystem responses to nutrient inputs, the targets for eutrophication can be said to be defined by Baltic Sea-wide or basin-specific ecosystem properties and responses to nutrient enrichment. The target setting is based on the analysis of data taking dose-responses, resilience and, in theory, also thresholds into account.

In principle, there may be several specific cases for setting the target: (1) a gradual change and response to nutrient enrichment, as well as a linear recovery when loads are reduced (Fig. 1.5A); (2) an abrupt change and response to nutrient enrichment, as well as a delayed recovery when loads are reduced (Fig. 1.5B); and (3) a gradual change and response to nutrient enrichment, as well as a linear recovery, but with a shift in baseline, when loads are reduced (Fig. 1.5C). Further, the combination of a threshold (Fig. 1.5B) and a shifting baseline (Fig. 1.5C) is a specific case (4) with an abrupt change and response to nutrient enrichment, as well as a delayed recovery including a shift in baseline, when loads are reduced (Fig. 1.5D). In cases (1) and (2) (Fig. 1.5A,C) there are no Baltic Sea-wide or basin-specific dose-response relations or thresholds, but rather gradual or more subtle responses to nutrient enrichment. In such cases, target setting might become subjective involving also expert judgement. Taking into account that the objective has been to improve the scientific basis for eutrophication target setting, the iden-

Table 1.5 *Tentative normative definition of Good Environmental Status in regard to nutrients, water transparency, algal blooms, plants, animals and oxygen.*

	Unaffected by eutrophication	Affected by eutrophication
Nutrients	The concentrations of nitrogen and phosphorus are consistent with the basin-specific or sub-basin-specific reference conditions, or shows only slight signs of disturbance compared to the basin-specific or sub-basis-specific reference conditions.	The concentrations of nitrogen and phosphorus shows signs of moderate (or significant) disturbance compared to basin-specific or sub-basin-specific reference conditions.
Water transparency	Water transparency is consistent with the basin-specific or sub-basin-specific reference conditions, or shows only slight signs of disturbance compared to the basin-specific or sub-basis-specific reference conditions.	Water transparency shows signs of moderate (or significant) disturbance compared to basin-specific or sub-basin-specific reference conditions.
Algal blooms	Algal blooms occur at a frequency and intensity which is consistent with basin- or site-specific reference conditions, or shows only slight signs of disturbance compared to basin-specific or sub-basis-specific reference conditions.	Algal blooms occur at a frequency and intensity which are moderately (or significantly) elevated compared to basin-specific or sub-basin-specific reference conditions.
Plants and animals	All or most disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present.	A moderate number of the disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are absent.
	The levels of macroalgal cover and angiosperm abundance are consistent with undisturbed conditions.	The macroalgal cover and angiosperm abundance is moderately (or more) disturbed and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.
	The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions.	The level of diversity and abundance of invertebrate taxa is moderately outside the range associated with the basin-specific or sub-basin-specific conditions.
	All or most of the disturbance-sensitive taxa associated with undisturbed conditions are present.	Many of the sensitive taxa of the basin-specific or sub-basin-specific communities are absent. Taxa indicative of pollution are present.
Oxygen	Oxygen concentrations are consistent with the basin-specific or sub-basin-specific reference conditions, or shows only slight signs of disturbance compared to the basin-specific or sub-basis-specific reference conditions.	Oxygen concentrations show signs of moderate (or significant) disturbance compared to basin-specific or sub-basin-specific reference conditions.

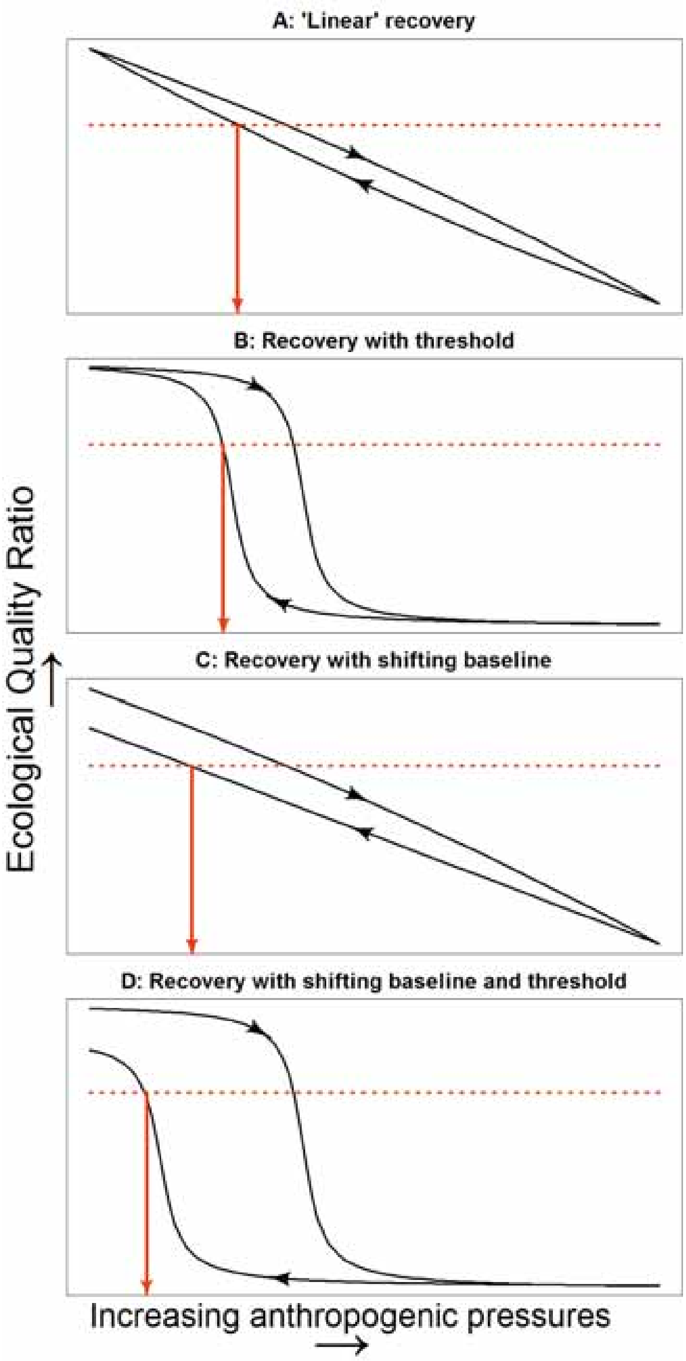


Figure 1.5 *Hypothetical models of the consequences of changes in anthropogenic nutrient loads on marine ecosystem quality. The dashed line indicates an environmental target for ecosystem quality; the red arrows indicate the estimated reductions in pressures needed to meet the target. The reductions increase from scenario A to D indicating that the fulfilment of the target in non-linear systems with a shifting baseline (cf. scenario D), e.g. caused by climate change or overfishing, calls for reductions significantly larger compared to linearly responding systems (cf. scenario A). Based on Duarte et al. (2009) and Kemp et al. (2010).*

tification of more or less linear time trends have been used to identify potential targets. Instead, as in cases (3) and (4) (Fig. 1.5B, D), the identification of statistically significant changes in ecosystem structure and functioning has been used to identify potential targets.

As the cause-effects relationships in regard to nutrient enrichment and eutrophication are well documented and widely acknowledged (Conley 2000, Vahtera et al. 2007, Conley et al. 2009, HELCOM 2009, Andersen et al. 2011), HELCOM TARGREV has focused the work on the identification of non-linearity and/or distinct ecosystem responses to nutrient inputs and nutrient enrichment in the Baltic Sea basins and sub-basins. By doing so, the work has applied the principles outlined in Fig. 1.5, in particular panel B and indirectly panel D. An added value of using this approach is that HELCOM TARGREV is indirectly sharing target setting principles with the Water Framework Directive, e.g. sub-division (here: 'basins', in the WFD: 'water bodies') and normative definitions. The key difference between the WFD, where the acceptable deviation is set by accepting a slight deviation from the reference conditions, and HELCOM TARGREV is that the eutrophication targets are based on change points where significant changes in structure and function have been identified. As a precautionary note, it should be emphasised that a change point in practice is equivalent to the target setting principle based on the reference conditions and acceptable deviations. Hence, the assessment of eutrophication status in the future will be possible with the currently used principles, methods and tools.

The methodologies for eco-region-wide and sub-eco-region-specific target setting developed by HELCOM TARGREV are regarded as a simple five-step target setting protocol, which is applied in Section 3.

2. Temporal trends for eutrophication indicators

The significance of any model, whether used for purely scientific or management purposes, relies on its ability to describe observations or derivations thereof. The confidence in a model further increases if the model is capable of describing variations over a large range of observations, typically in terms of variations in forcing as well as over time. To derive ecological targets, it is also important to describe the transition over time from a healthy ecosystem to an unhealthy one, as critical thresholds can be elucidated from such time series. In general terms, therefore, it is crucial to understand the past in order to predict for the future, i.e. a well-founded understanding of how the Baltic Sea deteriorated will provide important information to determine how to restore the ecosystem.

The objective of this chapter is to compile and collate various time series, obtained from simulation models and statistical analyses, that describe changes in the environmental factors (nutrient inputs and physical forcing) as well as response variables representing the five ecological objectives for eutrophication by HELCOM.

There are data from the Baltic Sea going back to the start of the 20th century, although these data are scarce, not sampled consistently and do not include all relevant variables. However, the early data can provide important information about the status of the Baltic Sea more than 100 years ago, a period believed to represent a Baltic Sea with minor disturbances from human activity. In this chapter, we will make use of all available data from the open parts of the Baltic Sea to reconstruct a time series, to the extent possible, of indicators representing the five ecological objectives of HELCOM. Metadata files showing the extent of data in time and space can be found on the HELCOM website (Folders» Monitoring and Assessment Group » CORESET/TARGREV » TARGETS 1/2012 » Station list for TARGREV report). For comparison, simulations from three dynamical models have produced hincasts for the same time span.

2.1 Nutrient inputs

The BONUS+ project ECOSUPPORT has reconstructed loads to the Baltic Sea from land and the atmosphere. The responsible scientists for this work are primarily: Oleg Savchuk and Bo Gustafsson at BNI, Stockholm; Kari Eilola at SMHI and Tuija Ruoho-Airola at FMI. The data set description is given in Gustafsson et al. (2012). Up until ca. 1970, the reconstruction is rather coarse and based primarily on population developments and assumptions on land use and industrial development (Fig. 2.1) following Savchuk et al. (2008) and Schernewski & Neumann (2005). The land loads are assumed to increase piecewise linearly, with a slow increase 1850-1950, and a more rapid increase after that. The land loads at 1900 correspond to the values given in Savchuk et al. (2008) and the increase to 1950 is found assuming dependence in proportion to population growth in major cities.

For the more recent period (1970-2006), the loads are compiled from data from the BED and PLC-5 for the riverine loads, and the direct point sources from HELCOM PLC reports and from Larsson et al. (1985) and references therein. A detailed description is given in Savchuk et al. (2012).

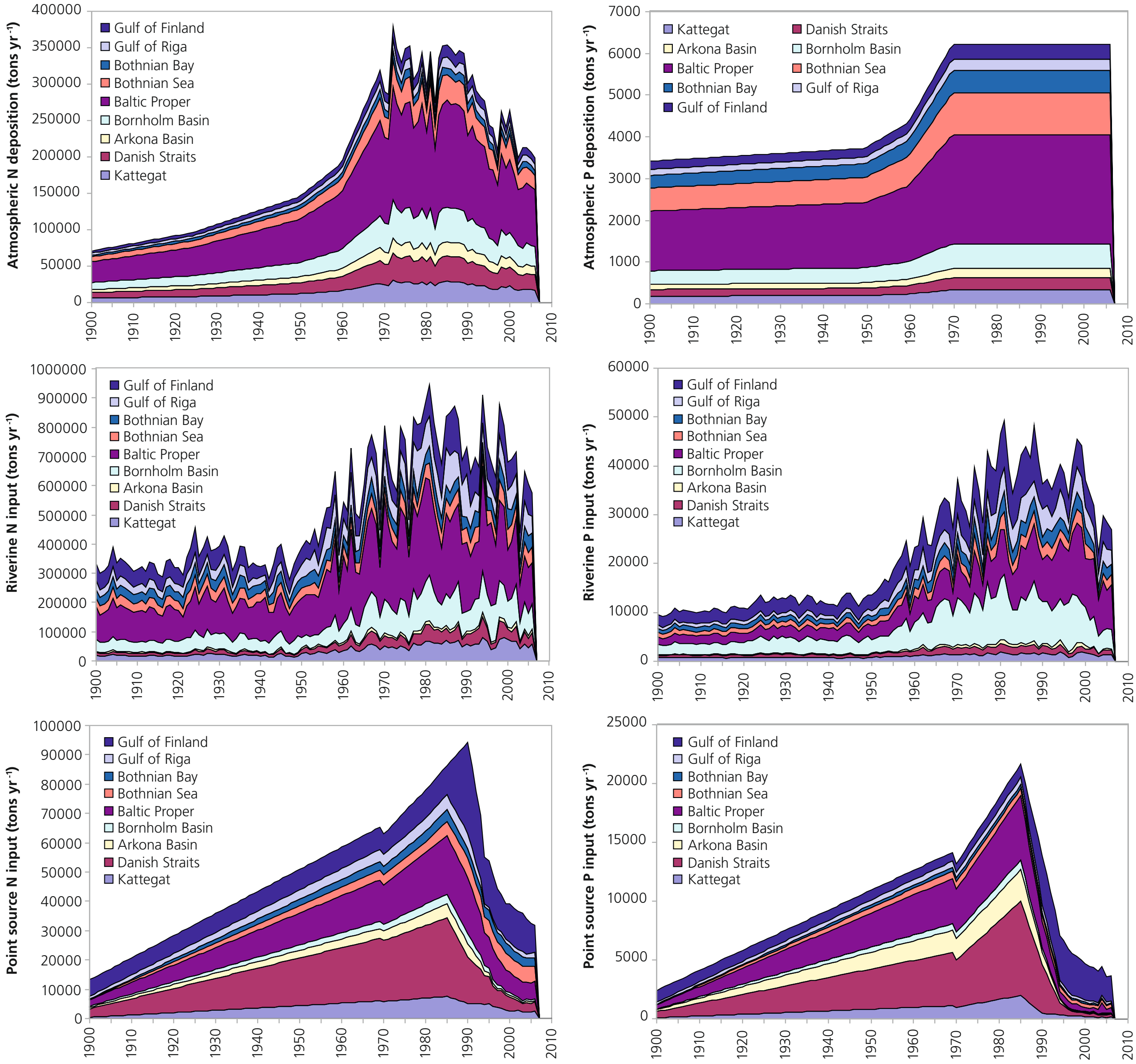


Figure 2.1 *Inputs of total nitrogen (left) and total phosphorus (right) from atmosphere (top), diffuse (middle) and point (bottom) sources to various basins in the Baltic Sea. Results from the ECOSUPPORT project.*

2.2 Nutrient and Chlorophyll *a* levels

In the HELCOM system of Ecological Objectives (EcoOs), nutrients and Chlorophyll *a* (Chl *a*) are directly linked to the EcoOs "Concentration of nutrients close to natural levels" and "natural levels of algal blooms", and both are HELCOM BSAP indicators for eutrophication. Nutrients and Chl *a* have subsequently been used as core indicators of eutrophication in the HELCOM integrated thematic assessment of eutrophication (HELCOM 2009) and the HELCOM Initial Holistic Assessment of the Ecosystem Health of the Baltic Sea (HELCOM 2010). In addition, nutrients and Chl *a* are relevant indicators of eutrophication describing good environmental status (GES Descriptor 5) in the Marine Strategy Framework Directive, as described in the Commission Decision 2010/477/EU.

Nutrient and Chl *a* concentrations in the water are important parameters for assessing the degree of eutrophication of marine habitats. Nutrients are causal agents of eutrophication, as increasing levels alter the ecosystem by directly stimulating fast growing autotrophic organisms, such as phytoplankton and free drifting algae (Krause-Jensen et al. 2008, Henriksen 2009). Through this, the nutrient concentration of the water indirectly affects the benthic vegetation as the increased amount of phytoplankton, of which Chl *a* is a measure, leads to an increased light attenuation in the water column and thereby reduces the main limiting factor, available light, at the sea bed. Another important effect of increased phytoplankton growth is the enhanced sedimentation of organic material, which may lead to both increased shading by settling on the vegetation (Krause-Jensen et al. 2008) and anoxia through increased oxygen consumption during decomposition (Conley et al. 2009).

2.2.1 Materials and methods

Nutrients and Chl *a* concentrations used in the present analysis were extracted from the Data Assimilation System (DAS), developed and hosted by the Baltic Nest Institute, Stockholm Resilience Centre, Stockholm University. DAS is a distributed database allowing access to databases hosted in Denmark, Finland, Germany and Sweden, containing hydrographical and chemical data for the Baltic Sea (Sokolov & Wulff 2011). However, for Chl *a* DAS is not complete and the data were supplemented by data collected for the EUTRO-PRO project and HELCOM Indicator Fact Sheets (Flemming-Lehtinen et al. 2008). Dissolved inorganic nitrogen (DIN) was calculated as the sum of ammonia, nitrate and nitrite, although if ammonia was missing DIN was approximated as the sum of nitrate and nitrite, since ammonia concentrations in the open surface waters are generally low.

The data were coupled with information about basins as defined in the BALTSEM model (Gustafsson 2000) and classified as either coastal or offshore areas according to the definition used by HELCOM, i.e. one nautical mile outwards from the baseline as defined in the WFD (see note on the cover page regarding the missing German data). Only positions classified as offshore were used in the analyses; surface waters, used for characterising nutrient levels, were defined as 0-10 m in the Kattegat and Danish Straits and 0-20 m in the Arkona Basin, Baltic Proper, Bornholm Basin, Bothnian Bay, Bothnian Sea, Gulf of Finland and Gulf of Riga data. These depth definitions represent the upper mixed layer in the open waters above the haloclines, which are situated at different depths in the Baltic Sea basins, although deeper than the depth definitions above. For Chl *a*, 0-10 m was used to represent the surface layer. In the open waters, this definition includes the upper mixed and productive layer.

The data set from DAS contained more than five million records with observations of varying quality across time. Given the amount of data, it was not possible to quality check observations individually and therefore an automated procedure was employed. For nutrients, known to display some degree of co-variation, outliers in the dataset were identified by first applying the Blocked Adaptive Computationally-Efficient Outlier Nominators (BACON) algorithm for multivariate covariance estimation, as implemented in the R-package "RobustX" (Stahel & Maechler 2009) for each basin followed by a visual inspection of the data. For other parameters, observations outside the 99% confidence interval for the distribution were identified and the data visually inspected.

Statistical model

The monitoring data underlie three main sources of variation that must be addressed in a combined analysis. For all variables, there are significant spatial gradients, significant seasonal patterns and significant interannual variations. The aim of the statistical analysis described here is to separate these different components to produce trends that are unbiased by differences in the seasonal and spatial sampling across the years. The general approach is described in Carstensen et al. (2006). Resolving spatial gradients and seasonal variations in the trend analysis is an advance to averaging observations over an area and seasonal window since more precise and unbiased estimates are produced (Carstensen 2007).

The measured nutrient and Chl *a* concentrations were first log-transformed before a Generalized Linear Model (GLM) was employed, separating the variation in the measurements into spatial variation (station), seasonal variation (month) and yearly variation (year). The model was parameterized using the GLM procedure in the statistical software package SAS/STAT 9.2 (SAS 2009). The station-specific means were then used to fit two Generalized Additive Models (GAM) containing a bivariate thin-plate spline describing the spatial variation as a function of the stations' geographic coordinates (in UTM projection 34), one covering the basins Kattegat and Danish Straits, and one covering the Arkona Basin, Baltic Proper, Bornholm Basin, Bothnian Bay, Bothnian Sea, Gulf of Finland and Gulf of Riga. The two models were parameterized using the GAM procedure in SAS/STAT 9.2 (SAS 2009).The estimates from the spatial model were then used to remove the spatial variation in the data by subtracting the spatial model component from the estimates. After spatial de-trending, a GLM only containing the temporal effects 'year' and 'month' was fitted for each basin in order to allow differences in trends and seasonal patterns across the basins.

The statistical approach above was applied to produce annual means for nutrients and Chl *a*, as well as winter levels for nutrients (Dec-Jan) and summer levels for Chl *a* (Jun-Sep). It should be stressed that these annual means represent the mean of the entire spatial division for which they were estimated; however, trends and targets can be calculated for any sub-division based on the estimated spatial distribution. Overall, the estimation of the seasonal variation within the model has been found to increase the precision of the estimation of yearly as well as seasonal means (Carstensen 2007, HELCOM 2009). The plots below the annual and seasonal trends have been scaled using separate axes since there can be differences in the ranges for some variables. These seasonal windows employed are in accordance with the procedures in HELCOM (2009). In this section, results are shown for the Baltic Proper only, whereas the results from the other basins are presented in Annex B. Although the trends are the main interest in this section, the seasonal and spatial variations are also presented to illustrate the soundness of the approach.

2.2.2 Results

Total nitrogen

The estimated spatial component of total nitrogen (TN) showed that nitrogen is highly unevenly distributed in the Baltic Sea, reaching concentrations of above 30 mmol l^{-1} in parts of the Gulf of Finland and the Gulf of Riga, while more open areas had concentrations of about half this level (Fig. 2.2). The spatial pattern was consistent with the major sources for nutrient inputs to the Baltic Sea. The spatial distributions for annual and winter means were similar.

The long term variation in the yearly values of TN showed that, despite strong year-to-year fluctuations, nearly all basins have experienced increasing levels of TN up to the late 1980s, at which point the rate of increase levelled out and even decreased for the Kattegat and part of the Danish straits (Fig. 2.3, Fig. B.1). It should also be stressed that there could be potential measurement problems with some of the earlier data, resulting in high values for data before 1970. Trends in the annual and winter TN were similar across all basins; however, the uncertainty of the winter means were about twice the annual means due to less data used for estimating the means.

The seasonal variation in TN showed a small decrease for most basins in the TN concentrations during the productive period (Fig. 2.4, Fig. B.2) beginning in spring (April) and ending again in autumn (October-November), mainly caused by the export of particulate organic matter from the pro-

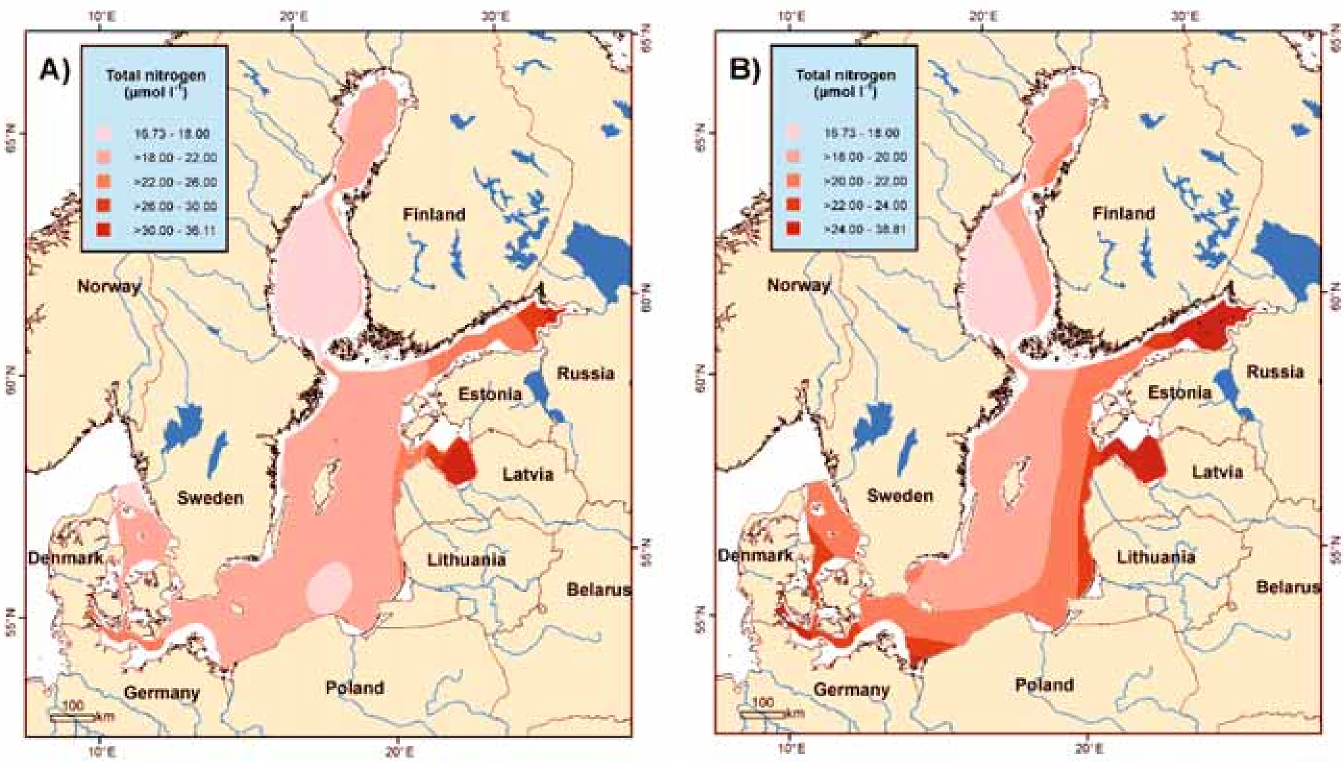


Figure 2.2 *Spatial variations in surface TN concentrations in the Baltic sea (0-10 m for Kattegat and Danish Straits; 0-20 m for others) estimated from the GAM approach. A) Annual mean distribution and B) winter mean distribution (Dec-Feb) represent 1968-2010 and 1970-2010, respectively (cf. Fig. 2.3).*

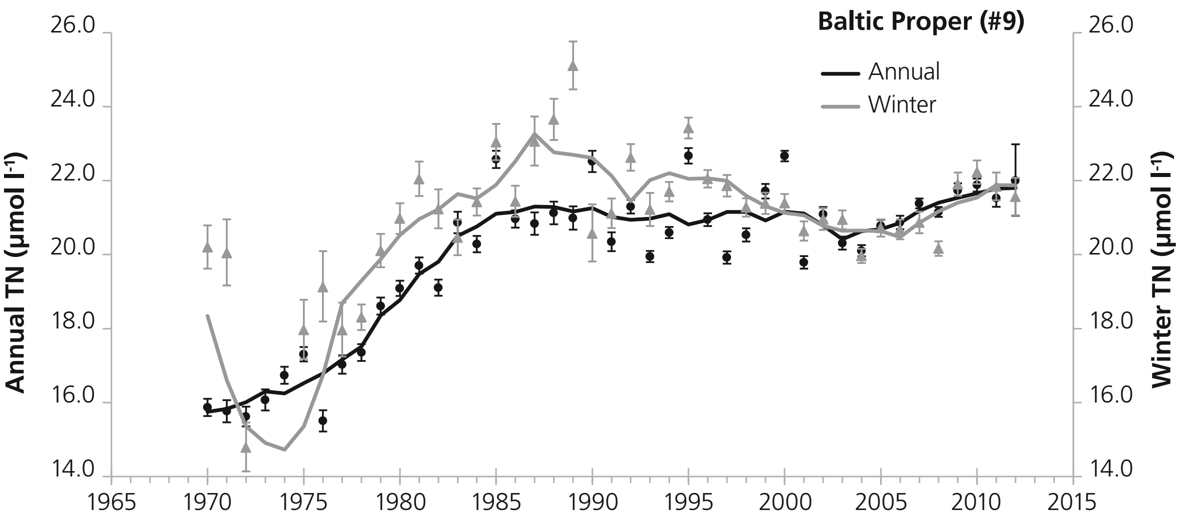


Figure 2.3 *Long-term trend in annual (black) and winter (grey) surface TN concentrations in the Baltic Proper (0-20 m). Lines indicate the five-year moving average (starting from 1970); error bars represent 95% confidence limits of the means. Other basins are shown in Annex B (Fig. B.1)*

ductive layer. In the Baltic Proper and the Arkona and Bornholm Basins, however, there was TN enrichment during July-August, which is most likely due to nitrogen fixation by cyanobacteria. The seasonal variation is also consistent with winter TN means being slightly higher than the annual means (Fig. 2.3).

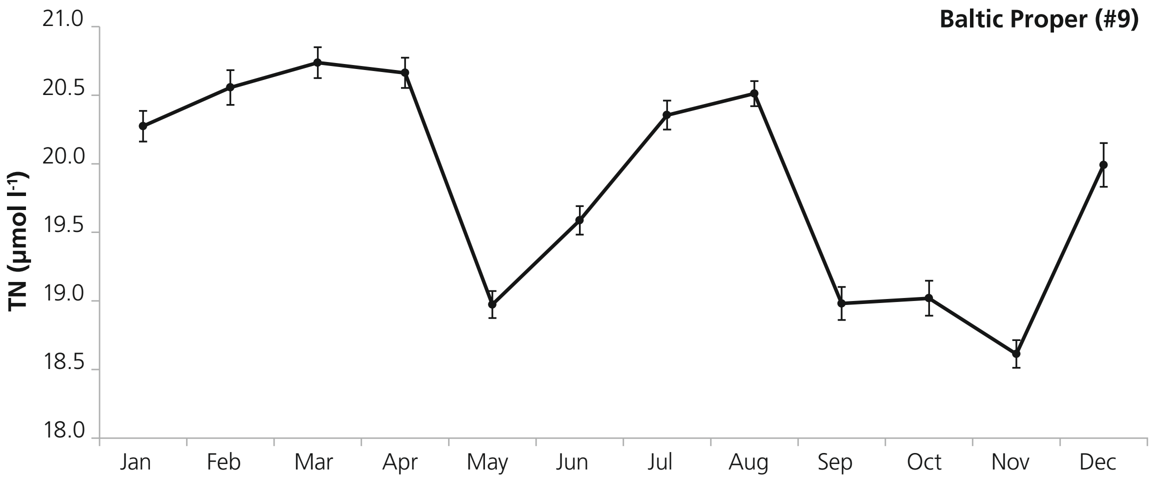


Figure 2.4 *Seasonal variations in the mean surface TN concentrations in the Baltic Proper (0-20 m) for the period 1968-2010 (cf. annual means in Fig. 2.3). Error bars represent 95% confidence limits of the means. Other basins are presented in Annex B (Fig. B.2).*

Dissolved inorganic nitrogen

The spatial variation in DIN concentrations showed the same pattern as for TN, with the highest concentrations in semi-enclosed areas such as the Gulf of Finland and the Gulf of Riga and through the Danish Straits; concentrations in open parts like the Baltic Proper, however, were lower (Fig. 2.5). The spatial pattern is consistent with what would be expected based on the major sources of nitrogen inputs, except for the Bothnian Bay where inputs are small. In the Bothnian Bay, DIN levels are also high because phosphorus is limiting algal production leading to excess DIN (non-depleted levels) throughout most of the productive season; and as the productive season is relatively short, DIN therefore remains high throughout extended periods of the year (Fig. B.4). The spatial distributions for annual and winter means were similar.

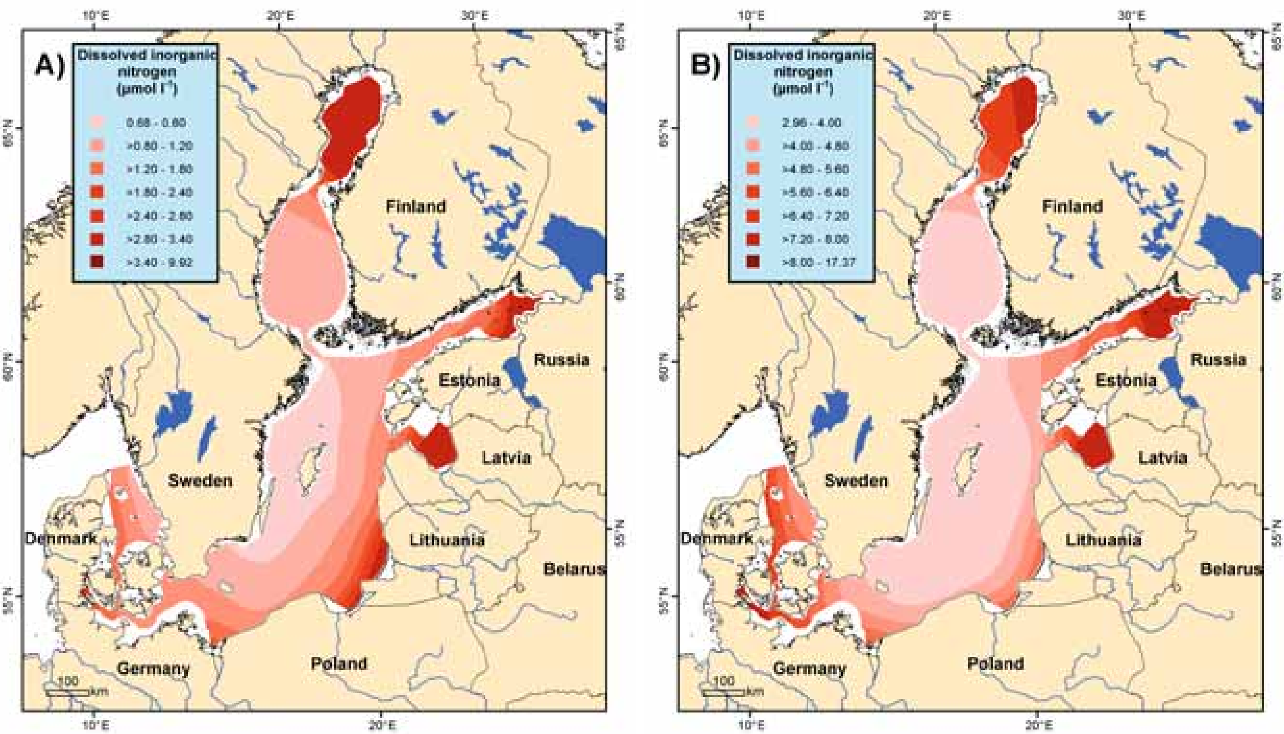


Figure 2.5 *Spatial variations in surface DIN concentrations in the Baltic sea (0-10 m for Kattegat and Danish Straits; 0-20 m for others) estimated from the GAM approach. A) Annual mean distribution and B) winter mean distribution (Dec-Feb) represent 1968-2010 and 1970-2010, respectively (cf. Fig. 2.6).*

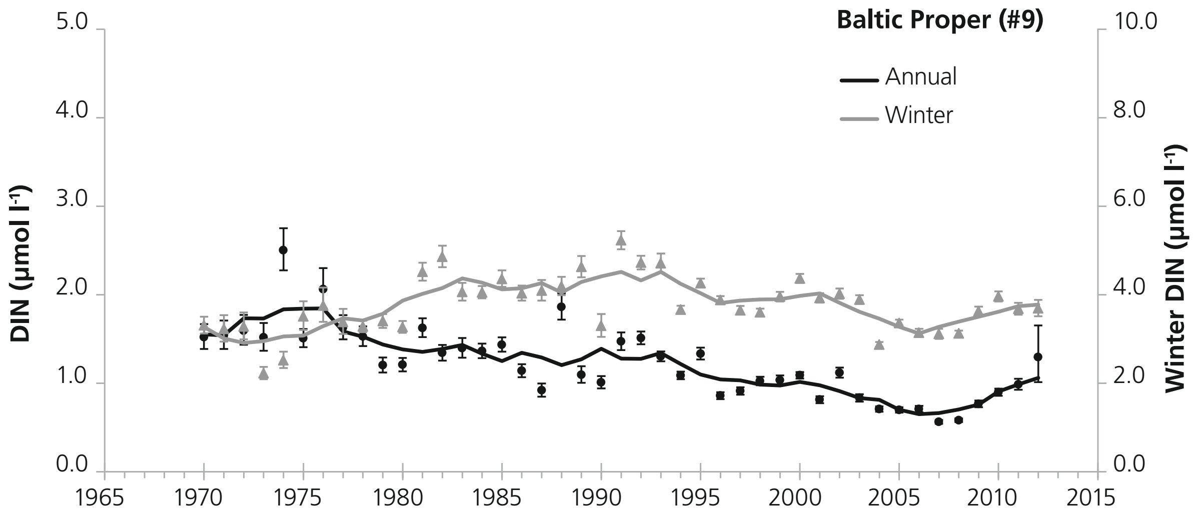


Figure 2.6 *Long-term trend in annual (black) and winter (grey) surface DIN concentrations in the Baltic Proper (0-20 m). Lines indicate the five-year moving average and error bars represent 95% confidence limits of the means. Other basins are shown in Annex B (Fig. B.3)*

The long-term temporal trends in DIN showed larger variation between years than TN (Fig. 2.6; Fig. B.3). To some extent, the pattern is similar to long-term changes in TN levels as the concentrations of DIN increases until the mid-1980s, after which DIN levels in several basins declined (particularly in the south-western Baltic Sea). Declines were larger for the annual means than for winter means, which could be due to extended productive seasons associated with the the warming trends of the Baltic Sea. Trends and seasonality are consistent with Nausch et al. (2008) and HELCOM (2009). Winter means were about 50% more uncertain than annual means due to less data used for their calculation.

The seasonal variation in DIN concentrations showed a marked seasonal pattern with the highest levels measured in December – March, with DIN being almost depleted during the summer (Fig. 2.7; Fig. B.4). This pattern is typically observed for DIN concentrations in mid-latitude marine systems and is explained by the accumulation during winter and the subsequent uptake of nitrogen by phytoplankton during spring and summer (Nausch & Nausch 2006, Nausch et al. 2008).

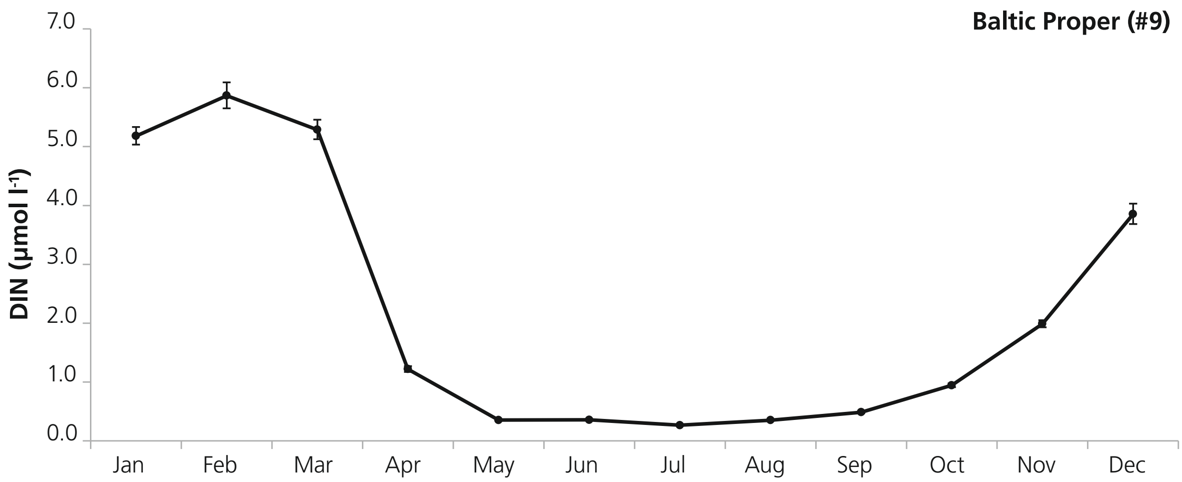


Figure 2.7 *Seasonal variations in the mean surface DIN concentrations in the Baltic Proper (0-20 m) for the period 1968-2010 (cf. annual means in Fig. 2.6). Error bars represent 95% confidence limits of the means. Other basins are presented in Annex B (Fig. B.4).*

Total Phosphorus

The estimated spatial distribution of total phosphorus (TP) concentration showed high concentrations of about 0.9 mmol l^{-1} in the Danish Straits, Gulf of Finland and Gulf of Riga, while large parts of the Baltic Proper have levels around 0.5 mmol l^{-1} (Fig. 2.8). This gradient further continued into the Bothnian Sea and Bothnian Bay, where the lowest levels were reached. The spatial distributions for annual and winter means were similar.

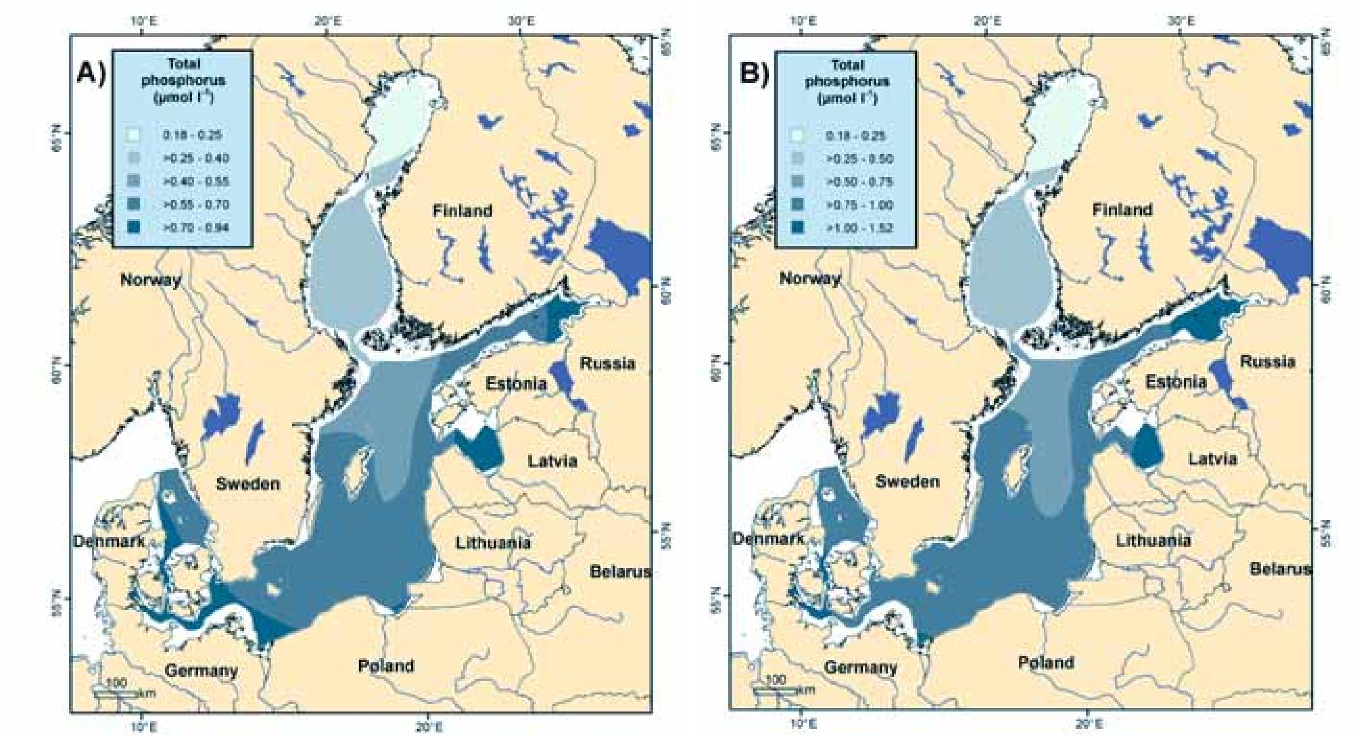
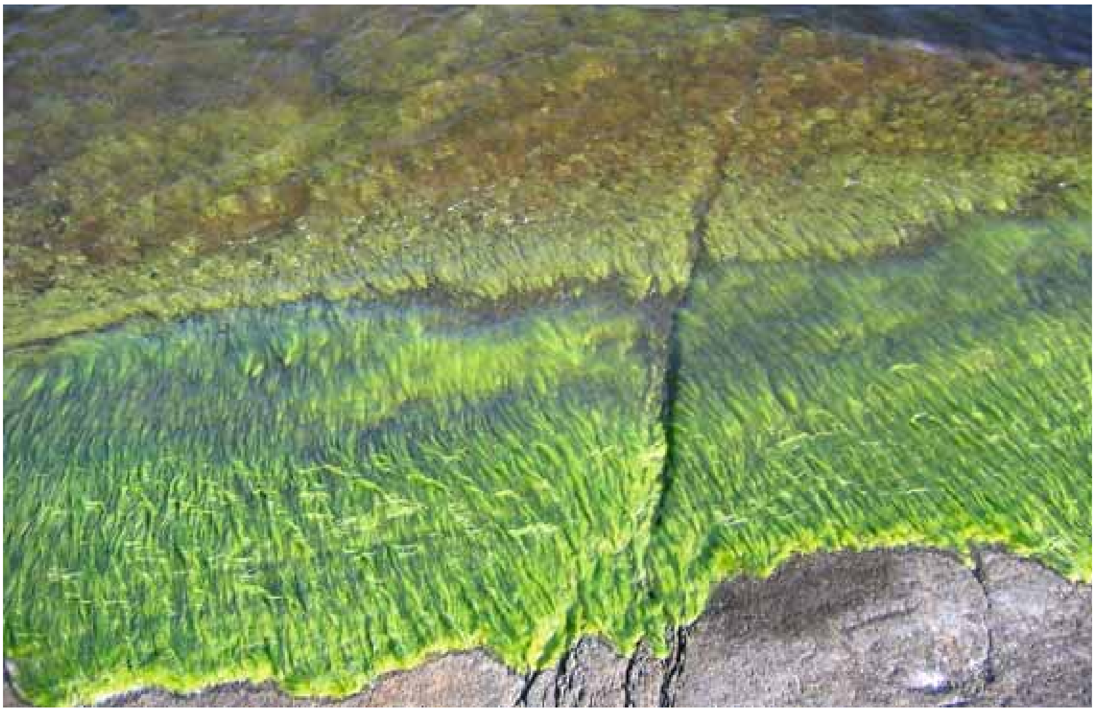


Figure 2.8 *Spatial variations in surface TP concentrations in the Baltic sea (0-10 m for Kattegat and Danish Straits; 0-20 m for others) estimated from the GAM approach. A) Annual mean distribution and B) winter mean distribution (Dec-Feb) represent 1967-2010 (cf. Fig. 2.9).*



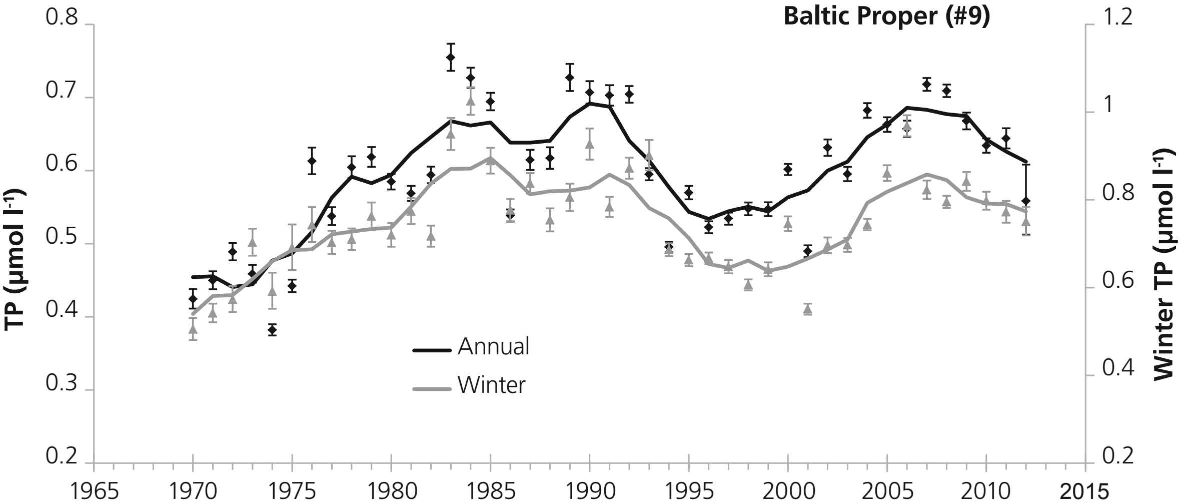


Figure 2.9 *Long-term trend in annual (black) and winter (grey) surface TP concentrations in the Baltic Proper (0-20 m). Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means. Other basins are shown in Annex B (Fig. B.5)*

The long term variation in the yearly TP concentrations showed a general increase with overlaying cyclic oscillations - probably connected to the trends in hypoxia and releases of phosphorus from sediments - in the Baltic Proper, the Arkona Basin, Bornholm Basin and partly the Bothnian Sea (Fig. 2.9; Fig. B.5). In the Danish Straits and the Kattegat, there has been a general decrease in TP levels since the mid-1980s, although a slight increase during the last five years is also apparent, probably connected to the trends in the central Baltic Sea. In the Gulf of Riga, TP levels declined slightly after 1990, whereas TP levels have remained high in the Gulf of Finland. It should also be stressed that there could be potential measurement problems with some of the earlier data, resulting in high values before 1970. Winter means were about twice as uncertain as the annual means due to less data used for their calculation.

The monthly estimates of TP had a unimodal pattern with lower concentrations during the summer months, typically between April and October (Fig. 2.10; Fig. B.6). Only the Bothnian Bay did not have a pronounced seasonal pattern. Moreover, there were differences in the seasonal patterns showing that the sedimentation of particulate organic matter following the spring bloom comes earlier in the south-western basins and later to the north.

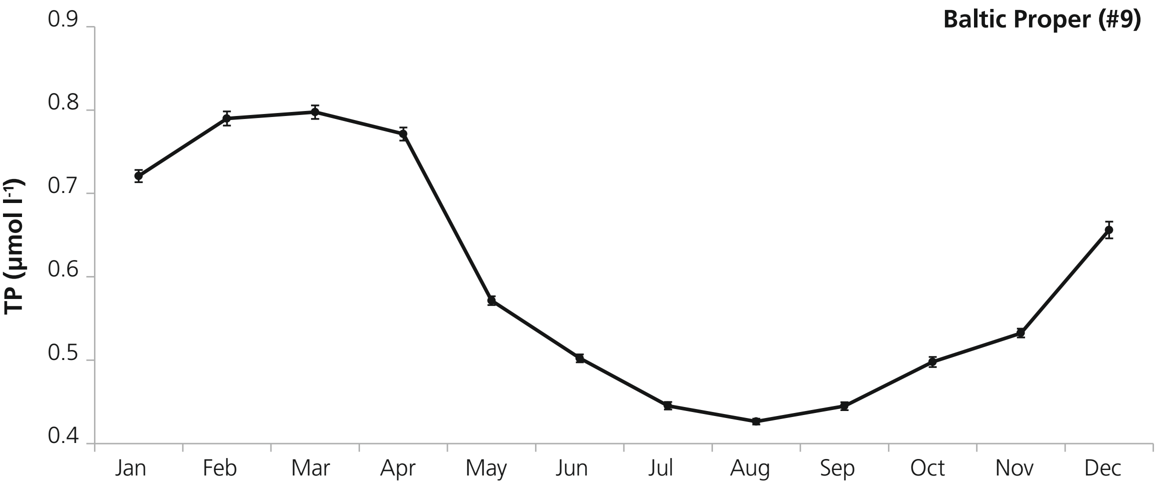


Figure 2.10 *Seasonal variations in the mean surface TP concentrations in the Baltic Proper (0-20 m) for the period 1967-2010 (cf. annual means in Fig. 2.9). Error bars represent 95% confidence limits of the means. Other basins are presented in Annex B (Fig. B.6).*

Dissolved inorganic phosphorus

The spatial pattern in dissolved inorganic phosphorus (DIP) was similar to that for TP with the highest concentrations of up to 0.3 mmol l^{-1} found in the western part as well as in the Gulf of Finland and the Gulf of Riga (Fig. 2.11). The spatial distributions for annual and winter means were comparable.

The long-term trends in DIP showed large year-to-year fluctuations; for most basins, however, there was a general increase in the concentration of DIP up until the end of the 1980s, after which the concentrations start to decrease (Fig. 2.12; Fig. B.7). Nonetheless, in the most recent years, DIP levels have increased in the Baltic Proper and

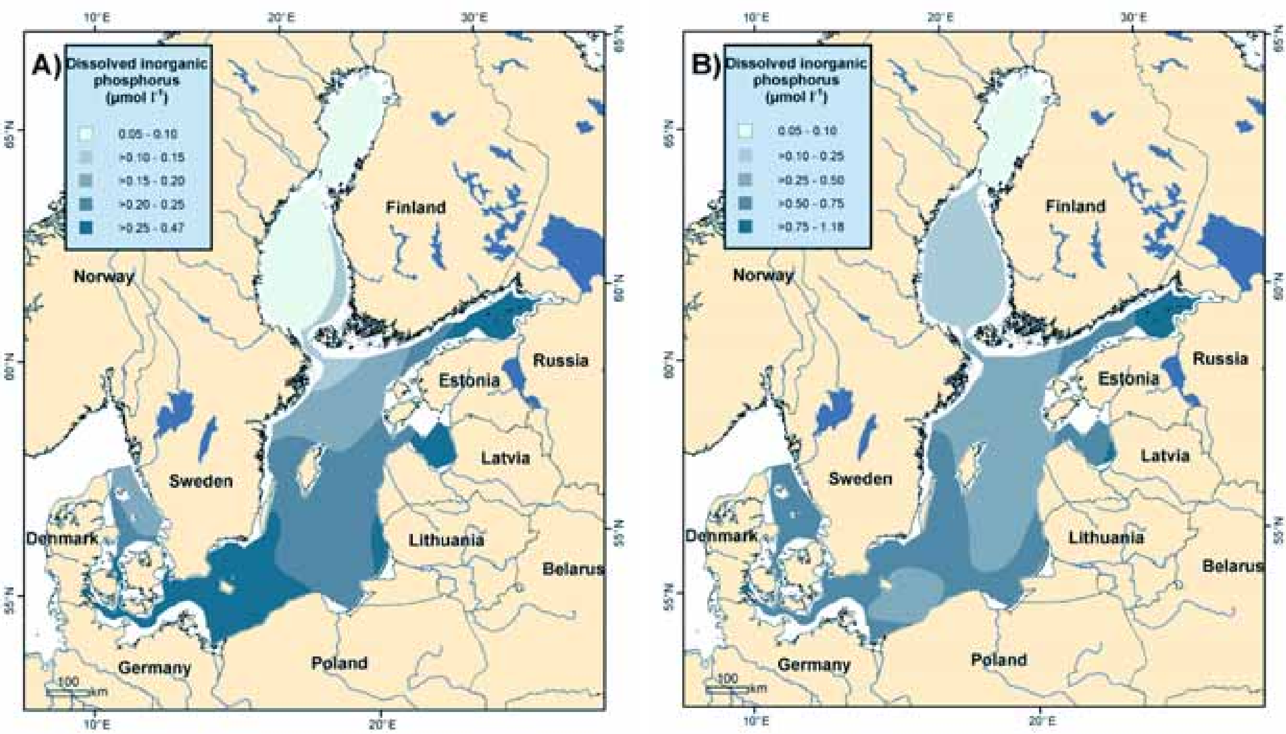


Figure 2.11 *Spatial variations in surface DIP concentrations in the Baltic sea (0-10 m for Kattegat and Danish Straits; 0-20 m for others) estimated from the GAM approach. A) Annual mean distribution and B) winter mean distribution (Dec-Feb) represent 1967-2010 and 1968-2010, respectively (cf. Fig. 2.12).*

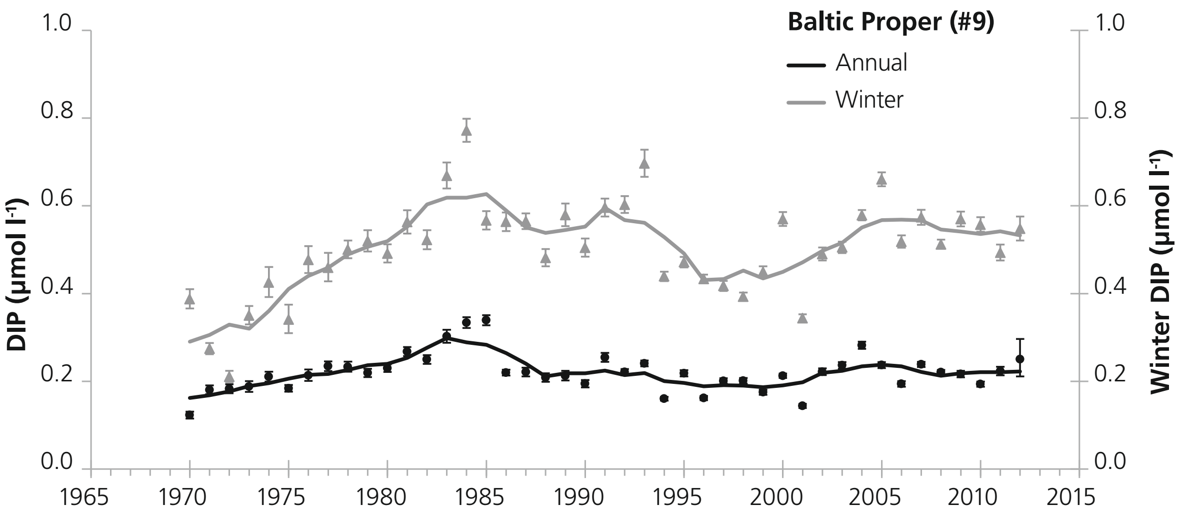


Figure 2.12 *Long-term trend in annual (black) and winter (grey) surface DIP concentrations in the Baltic Proper (0-20 m). Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means. Other basins are shown in Annex B (Fig. B.7)*

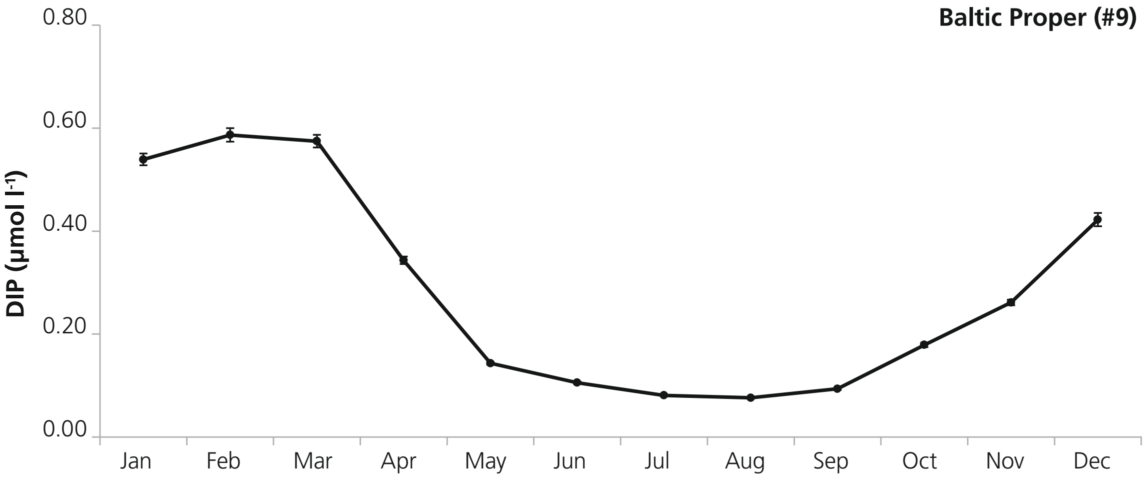


Figure 2.13 *Seasonal variations in the mean surface DIP concentrations in the Baltic Proper (0-20 m) for the period 1967-2010 (cf. annual means in Fig. 2.12). Error bars represent 95% confidence limits of the means. Other basins are presented in Annex B (Fig. B.8).*

the Bornholm and Arkona Basins in response to the hypoxia-enhanced release of phosphorus from the sediments (Conley et al. 2002, 2009, see also Section 2.4). DIP levels decreased in the Kattegat and the Danish Straits during the 1990s and have remained constant since then. Some declines have also been observed in the Gulf of Riga and the Gulf of Bothnia, but not in the Gulf of Finland. Overall, the winter and annual means display similar trends, but winter means were about 50% more uncertain than annual means due to less data used for their calculation.

The seasonal variation in DIP followed the same pattern as DIN with a strong decline in concentrations (Fig. 2.13, Fig. B.8), nearly reaching depletion during the summer months due to an uptake by autotrophic organisms (Nausch & Nausch 2006). In the Bothnian Bay and partly the Bothnian Sea, DIP remained at low levels throughout the entire annual cycle (Annex B). DIP levels generally start increasing earlier than DIN due to the faster recycling of DIP (Nausch et al. 2008).

Chlorophyll a

The spatial variation in Chlorophyll *a* (Chl *a*) indicated that concentrations closely follow the distribution of nutrients, with the highest concentrations occurring in the Gulf of Finland, the Gulf of Riga and along the southern and eastern coasts of the Baltic Sea (Fig. 2.14). The annual and summer spatial distributions were similar.

The yearly trends in Chl *a* showed increases across all basins, despite stagnating or even decreasing nutrient levels (Fig. 2.15, Fig. B.9). In the Baltic Proper and the Bornholm and Arkona Basins, there has been an almost doubling of the Chl *a* levels over the monitoring period. This apparent paradox is consistent with similar observed tendencies in coastal ecosystems worldwide and has been

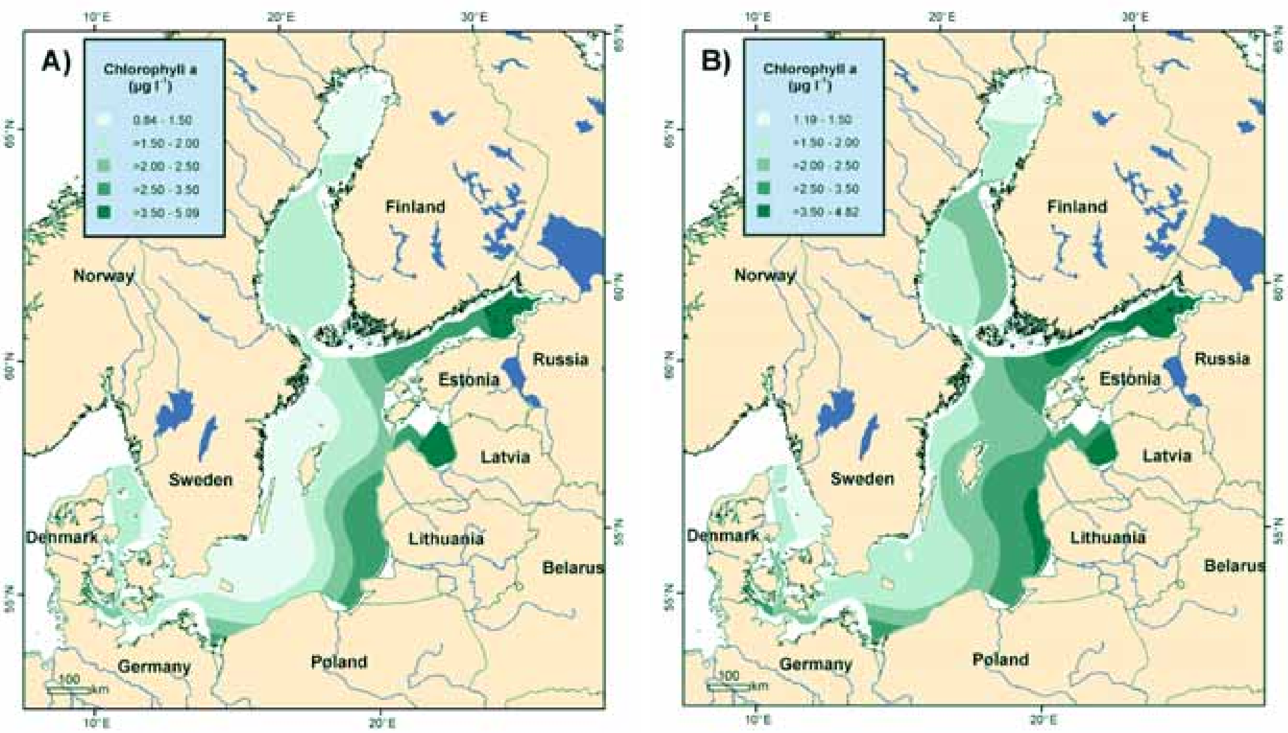


Figure 2.14 *Spatial variations in surface Chl a concentrations in the Baltic sea (0-10 m) estimated from the GAM approach. A) Annual mean distribution and B) summer mean distribution (Jun-Sep) represent 1972-2010 (cf. Fig. 2.15).*

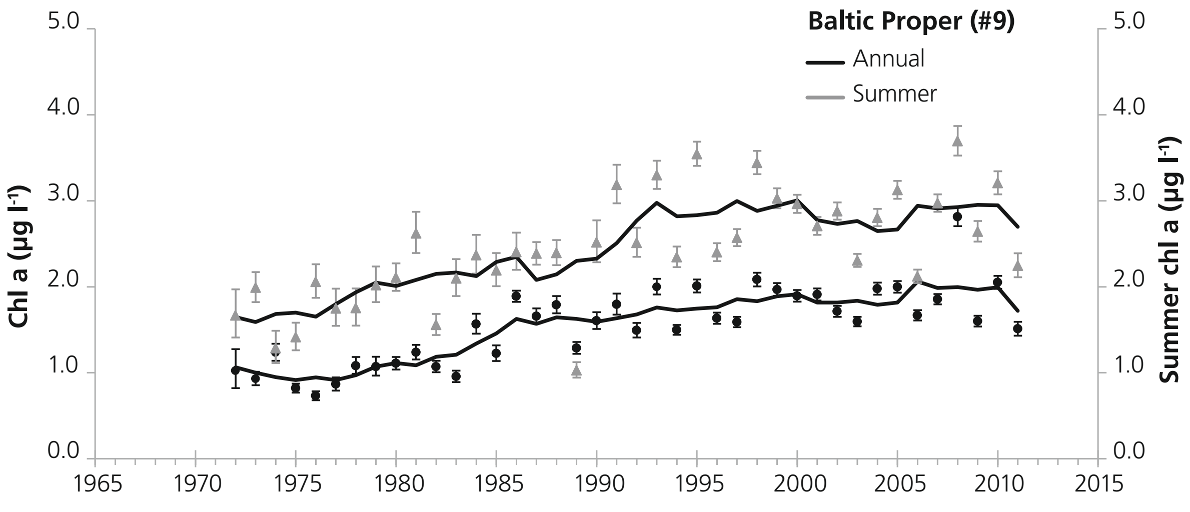


Figure 2.15 *Long-term trend in annual (black) and summer (grey) surface Chl a concentrations in the Baltic Proper (0-10 m). Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means. Other basins are shown in Annex B (Fig. B.9)*

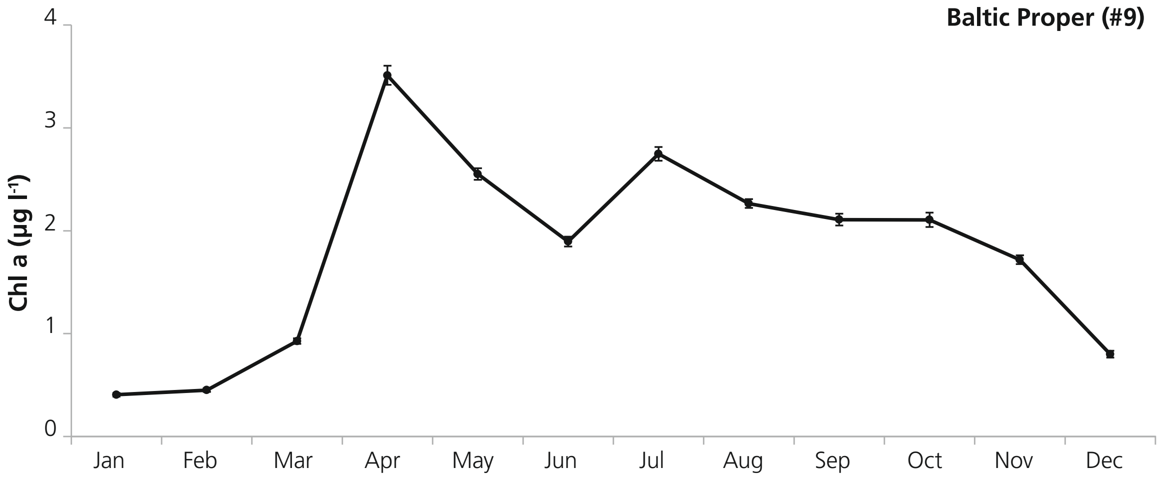


Figure 2.16 *Seasonal variation in the mean surface Chl a concentrations in the Baltic Proper (0-10 m) for the period 1972-2010 (cf. annual means in Fig. 2.15). Error bars represent 95% confidence limits of the means. Other basins are presented in Annex B (Fig. B.10).*

attributed to climate changes and altered food-webs (Duarte et al. 2009, Carstensen et al. 2011). Summer and annual means showed similar trends, although the summer means were almost twice as uncertain as the annual means for most basins. It should be noted that in the Bothnian Bay, both summer and annual means were almost identical since there were few observations outside the summer seasonal window (Jun-Sep, see Fig. B.10).

The seasonal variation in Chl *a* showed peaks for most basins in spring and late autumn (Fig. 2.16, Fig. B.10) representing the two main phytoplankton blooms occurring in the Baltic Sea (Nausch & Nausch 2006, Wasmund & Siegel 2008). In the Baltic Proper, a summer peak associated with cyanobacteria blooms was also observed. However, the spring bloom was very pronounced in the Gulf of Finland and the Gulf of Riga, consistent with the larger amount of inorganic nutrients accumulated during winter in these two basins (cf. Fig. B.4 and B.8). The seasonality in the Bothnian Bay and Bothnian Sea was more unimodal, suggesting that most production took place during summer since the nutrient accumulation in winter was relatively low. A shift occurred in the timing of the spring bloom from March-April in the south-western parts of the Baltic Sea (Kattegat, the Danish Straits) to occurring during April-May in the north-eastern parts (Gulf of Finland and Gulf of Riga), reflecting the north-south climatic gradient through the Baltic Sea.

2.3 Secchi depth

In the HELCOM system of Ecological Objectives (EcoOs), water transparency is directly linked to the EcoO "clear water" and is one of the HELCOM BSAP indicators for eutrophication. It has subsequently been used as one of the core indicators of eutrophication in the HELCOM integrated thematic assessment of eutrophication (HELCOM 2009) and the HELCOM Initial Holistic Assessment of the Ecosystem Health of the Baltic Sea (HELCOM 2010). Moreover, it is one of the indicators of eutrophication describing good environmental status (GES Descriptor 5) in the Marine Strategy Framework Directive, as described in Commission Decision 2010/477/EU.

The white Secchi disc is one of the few early hydrobiological measuring devices still in use. In the Baltic Sea, observations have been made from the end of the nineteenth century to the present. Thus, Secchi data provide unique first-hand information on environmental changes in the Baltic Sea, starting from a time when it was in a near-pristine state. Secchi depth, a measurement of water transparency, has been linked to eutrophication. It indicates the attenuation of light penetrating into water, governed by its absorption and scattering properties.

Secchi depth has decreased in the Baltic Sea from the early 1900s to the present (Sandén and Håkansson 1996, Launiainen et al. 1989, Fleming-Lehtinen & Laamanen 2012). Preparing the first thematic assessment on eutrophication in the Baltic Sea, the HELCOM EUTRO and EUTRO-PRO project developed tentative reference conditions for Secchi depth based on data mining, modelling and expert judgement (HELCOM 2006, HELCOM 2009, Fleming-Lehtinen 2007). These tentative targets, however, were neither harmonized between basins nor with the targets of the other eutrophication indicators. In addition, the targets defined by acceptable deviations from the reference conditions were not scientifically justified. As a result, the targets set for Secchi depth are in need of revision.

The HELCOM TARGREV project has a two-step approach for setting the Secchi depth target: first, a spatio-temporal data-mining approach is used in order to define and revise the targets; and second, adjusting the targets is investigated through the optical properties of the water. The harmonization of the Chl *a* and Secchi depth target-setting is then investigated using the same approach.

2.3.1 Materials and methods

A data set containing Secchi depth measurements from the entire Baltic area dating back to 1903 was compiled from numerous sources, including the ICES database with observations from the entire Baltic Sea between the years 1903 and 2009 (see also Aarup 2002), complemented with the Finnish Institute of Marine Research (now SYKE Marine Research Centre) datasets. Data from the Swedish Meteorological and Hydrological Institute (SMHI) database SHARK, the Oceanographic database of IMWM in Poland, the Latvian Institute of Aquatic Ecology and the Centre of Marine Research in Lithuania received during the HELCOM EUTRO project were also included. The data were handled as described in Fleming-Lehtinen & Laamanen (2012) before further analysis.

The Secchi depth observations were decomposed into spatial variation, seasonal variation and temporal trends using the same statistical approach as described for nutrients and Chlorophyll *a* (Chl *a*), although Secchi depth was approximately normally distributed and therefore not log-transformed.

2.3.2 Results

The spatial variation in Secchi depth showed that the highest mean values, up to 10 m, were found in the open parts of the Baltic Sea, decreasing to almost half the depth when entering the gulfs and parts of the belt seas (Fig. 2.17). There were also reduced Secchi depths near major river outflows in the southern and eastern Baltic Sea. The annual and summer spatial distributions were similar.

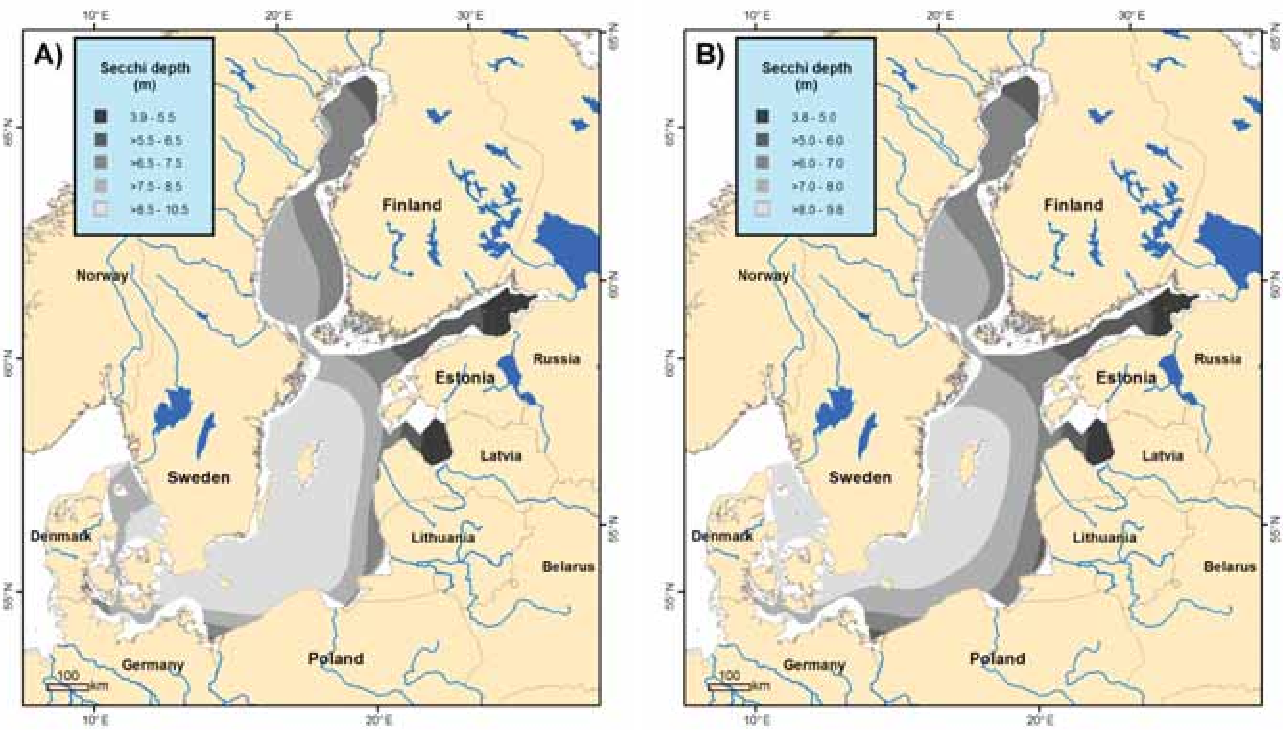


Figure 2.17 *Spatial variations in Secchi depth in the Baltic sea estimated from the GAM approach. Annual mean distribution (left) and summer mean distribution (right, Jun-Sep) represent 1903-2009 (cf. Fig. 2.18).*

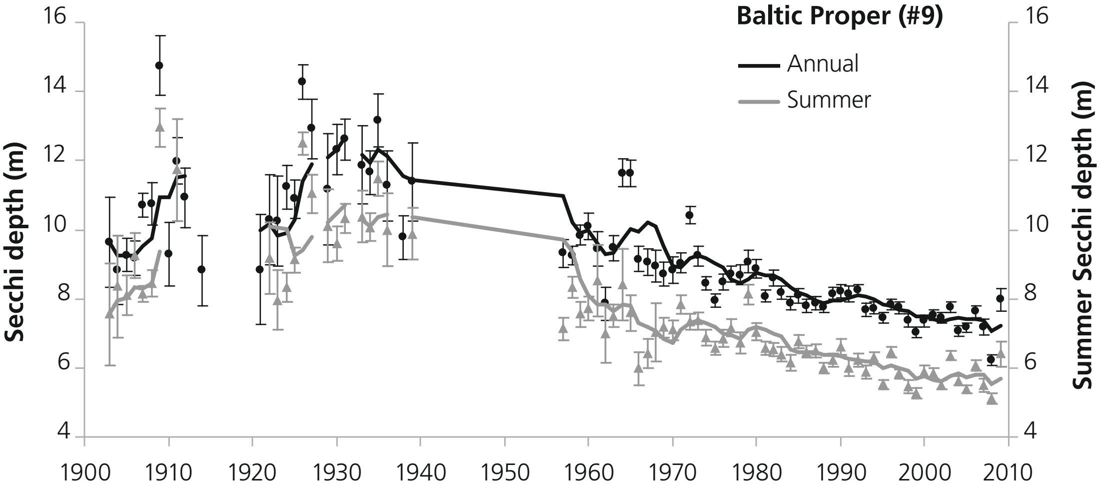


Figure 2.18 *Long-term trend in annual (black) and summer (grey) Secchi depth in the Baltic Proper. Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means. Other basins are shown in Annex B (Fig. B.11)*

The yearly estimates showed declining trends of Secchi depth over the last hundred years in all basins of the Baltic Sea (Fig. 2.18; Fig. B.11). In several basins, however, there was a tendency for the Secchi depth to stabilise after 1990, whereas in other basins such as the Gulf of Riga the decline continued. Long-term trends in annual and summer means were similar, and the uncertainty of annual means was only slightly better than the summer means (standard error of the means ~15-20% higher) suggesting that the two indicators are almost equally good from a statistical point-of-view.

The monthly estimates of Secchi depth did not have a consistent common pattern across all basins (Fig. 2.19; Fig. B.12). Some basins, such as the Arkona and Bornholm and the Baltic Proper, had higher transparency during the winter months and less during the summer months (June-September), when large algal blooms would occasionally form. In the Kattegat and the Danish straits, the Secchi depth was lower during the spring bloom period with a marked increased afterwards in the Danish straits (Annex A). In the Gulf of Bothnia, Gulf of Finland and Gulf of Riga, the seasonal variations were much less pronounced.

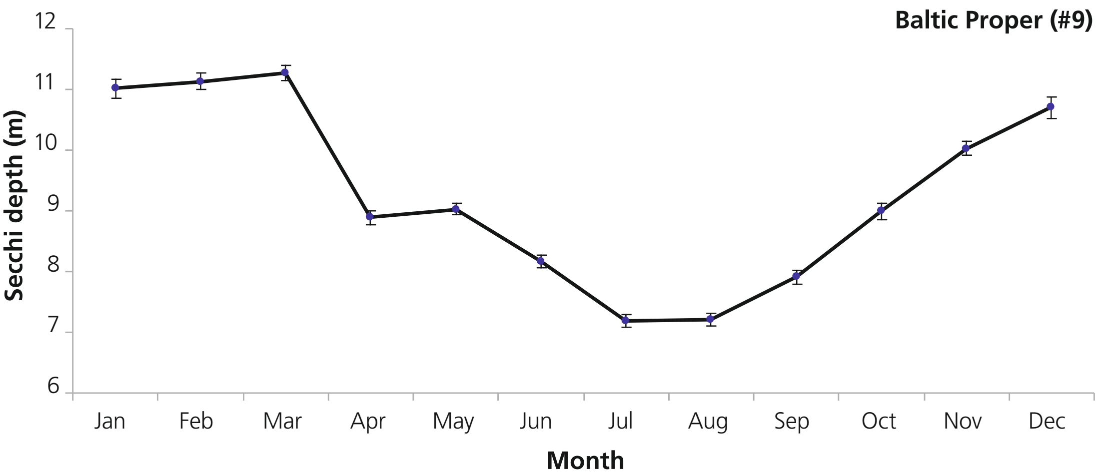


Figure 2.19 *Seasonal variations in the mean Secchi depth in the Baltic Proper for the period 1903-2010 (cf. annual means in Fig. 2.18). Error bars represent 95% confidence limits of the means. Other basins are presented in Annex B (Fig. B.12).*

2.3.3 Adjusting for other factors affecting Secchi depth

Eutrophication is defined as an increase in the supply rate of anthropogenically originated organic matter to an ecosystem (Nixon 1995). Despite the fact that allochtoneous organic matter may also be linked to eutrophication, it is relevant to separate the autochtoneous and nutrient-driven components of Secchi depth when using it as one of the eutrophication indicators. No significant straightforward relation between Secchi depth and individual water quality parameters have been observed in the Baltic Sea, under regular circumstances (Sandén & Håkansson 1996, Fleming-Lehtinen & Laamanen 2012, Stefan Simis *present data*). This is expected as explained below.

Secchi depth indicates water clarity and is related to the attenuation (absorption as well as diffusion by scattering) of sunlight penetrating into the water column (Preisendorfer 1986). In seawater, suspended particulate matter, chromophoric dissolved organic matter and living planktonic organisms, mainly phytoplankton, contribute to the attenuation of light. The absorption and scattering properties of light influence the vertical visibility of a submerged Secchi plate.

Only phytoplankton shows a direct causal relationship to anthropogenic nutrient loading in coastal systems, whereas CDOM may reside in the system for a long period, even in the absence of fresh depositions from terrestrial sources.
In order to use Secchi depth as an indicator of eutrophication, it is important to separate the contributions of these short and long-lived optical components. To accomplish this, an optical model of the system is required that relates the optical properties of the main water constituents to the fate of light in the sea.

The major contributors to absorption and/or scattering of light in the Baltic Sea water are CDOM and phytoplankton (Babin et al. 2003, Ferrari & Dowell 1998). The abiotic particles are not as strictly eutrophication-related; moreover, they are not assumed to be a major contributor to absorption and scattering in the open Baltic Sea (in coastal areas, the situation is more complex). The production of heterotrophic micro-organisms in relation to phytoplankton production is generally low in the marine environment, although some exceptions can be found in the Baltic Sea; in the Bothnian Bay the heterotrophic production may be as high as 2/3 of the phytoplankton production (Sandberg et al. 2004). Nevertheless, microscopic heterotrophic organisms are not expected to be a significant contributor to light attenuation as they are not dependant on photopigments. Finally, the absorption by pure water is significant but known from the literature (e.g. Buiteveld et al. 1994, Pope & Fry 1997).

The relation between algal biomass, described through Chl *a* and Secchi depth has been studied earlier in the open Baltic sub-basins (Fleming-Lehtinen & Laamanen 2012). In this work, based on Chlorophyll *a* and Secchi depth observations between 1972 and 2006, the median portion of phytoplankton of the total matter affecting Secchi depth was estimated between 13% and 17% in the summer, with great variation during the spring.

When revising and harmonizing the Secchi depth targets through bio-optical modelling, the HELCOM TARGREV project aimed to determine 1) whether there are grounds for re-estimating the targets set through data mining due to variations in attenuation caused by CDOM; and 2) whether the targets set for Chl *a* are in line with the targets set for Secchi depth. This was carried out through further investigating the relation of Secchi depth with the inherent optical properties of the Baltic Sea, especially in relation to the two most significant parameters CDOM and Chlorophyll *a*, using bio-optical modelling.

Methods

A bio-optical model was used to determine the effect of different concentrations of optically active substances (water, CDOM, phytoplankton) on the estimated Secchi disk depth. The model uses definitions of the specific inherent optical properties (absorption and scattering per unit of concentration) by CDOM, phytoplankton through Chl *a* and water, and the scattering of light by phytoplankton particles on the total attenuation of light in the water column. These definitions were obtained from SYKE's bio-optical data collected on r/v Aranda cruises during the spring and summer months between 2008 and 2011. Because such data are difficult to obtain, some generalizations had to be made: 1) the inherent optical properties of the phytoplankton are given as an average for either a spring or a summer phytoplankton population, without regard for regional trends in the phytoplankton community; and 2) the absorption spectrum of CDOM is expressed as a function of the absorption at a reference waveband and an exponential slope factor, where the latter is determined from the average relation between the two in the whole data set of optical properties of the Baltic Sea. While it is common in bio-optical studies to express CDOM absorption in this manner, regional variations in the slope factor are not taken into account.

Secchi depth was calculated with the radiative transfer solving software package Ecolight (version 5.0, Sequoia Scientific), which provides a numerical solution to the light field at discrete depths in the water column. The software applies a photopic function to the upwelling light field, from which the Secchi disk depth is simulated. In all results that follow, the radiative transfer model was run for the light field modelled for cloud-free conditions at noon in the central northern Baltic Proper, wind speed at 5 m/s and an infinite optical depth (i.e. no bottom effect).

Within the bio-optical data set of the Baltic Sea collected in 2008-2011, a deviation of modelled Secchi depth from *in situ* observed Secchi depth was found for Secchi depths deeper than 5.5 m (Fig. 2.20). This deviation is possibly caused by a difference in actual observation circumstances and the optimal modelled optical circumstances, or other natural differences between the actual and modelled optical conditions. We used a correction of the simulated results to match the expected observed results, as follows:

$$Z_{SD\ exp} = \frac{\mathit{ln} Z_{SD\ mod} - \mathit{ln}\ 1.2066}{0.2822},$$

where $Z_{SD\ exp}$ is the expected Secchi depth and $Z_{SD\ mod}$ is the Secchi depth simulated by the bio-optical model.

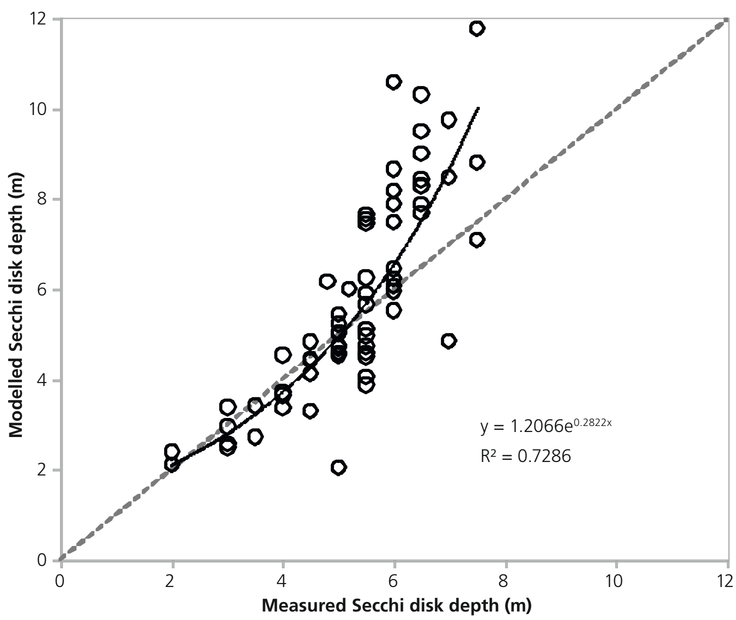


Figure 2.20 *Modelled vs. in-situ measured Secchi disk depth (n = 69). The modelled results are obtained with Ecolight 5 using in-situ vertical profiles of absorption, beam attenuation and a backscattering of light. The dashed line marks unity, whereas the unbroken line describes an exponential trend. The modelled results depart from unity only at Z_{SD} > 5.5 m.*

The data-mining-based targets for Secchi depth need to be evaluated regarding CDOM-caused changes and variations in the optical properties of the water. This work examined whether there were grounds for further adjusting the targets in areas where CDOM variation is substantial or the concentration has increased since the reference period. The deviation from the Secchi target caused by CDOM absorption was estimated separately for the summer (June-July) and spring (April-May) periods, due to the inter-annual differences in the bio-optical model. Because of the lack of sufficient long-term information on CDOM concentrations in the Baltic Sea water, the evaluation was based primarily on investigating the effects of the natural variation of CDOM concentration. The analysis was done separately for the sub-basins. The sub-basin-specific natural levels and variation of CDOM was achieved from previous work (Stedmon et al. 2000, Ylöstalo et al., *in prep.*), except for the Gulf of Riga which were excluded from the analysis. The average summer time CDOM was used to describe general CDOM absorption conditions, and the 97.5-percentile of CDOM concentration, achieved through adding double the standard deviation to the average, was used to describe conditions with unusually high, yet naturally possible levels of CDOM absorption.

The TARGREV long-term Chl *a* and Secchi depth data sets (see chapters on data mining for detailed descriptions) were used to investigate possible long-term changes in the optical properties of the water. Annual averages of the period of June-July were used in the comparison, in order to achieve best possible homogeneity in the values.

In order to investigate whether the targets of good environmental status set for Chl *a* are in line with the targets set for Secchi depth, the former were compared to simulated concentration of Chl *a* at the Secchi reference condition level. The Secchi depth targets were taken as the starting point to examine possible harmonization needs. Both targets are based on data mining, but the Chl *a* data extends only to a time period already affected by eutrophication, and could thus be considered scientifically less reliable than that of Secchi depth. The analysis was carried out separately for the spring and summer periods (as described above) using the sub-basin-specific average CDOM level. For comparison, the previous unrevised Chlorophyll *a* target for the summer period set by HELCOM EUTRO (HELCOM 2009) was also included, transforming all values back to the annual averages using the seasonal variation estimated in the data mining exercise (Fig. 2.19 and Annex B). The Gulf of Riga was excluded from the analysis due to insufficient information on the CDOM concentrations.

Results and discussion

Secchi depth decreases exponentially with added scattering or absorbing matter in the water (Fig. 2.21). In the range of absorption by CDOM around the median for open Baltic Sea waters (0.8 - 1.5 m^{-1} at 375 nm), the theoretical maximum Secchi depth lies in the order of 13 m, if no particulate matter is present. Within this range of CDOM concentration and at Chlorophyll *a* levels below 20 mg m^{-3}, Secchi depth varies from 11 m in the summer and 7 m in the spring depending on the phytoplankton biomass. At the higher limit of CDOM absorption, as could be observed for example in the Neva Bay, Secchi depth in the absence of phytoplankton and other particles would never exceed five meters. At the lower limit of CDOM absorption, only water contributes to the attenuation and scattering of light and we then find that Secchi depth corresponds to the values measured in oceanic waters, around 16 m (Fig. 2.21). In these circumstances, Secchi depth corresponds strongly to Chlorophyll *a* concentration, especially at low levels, and may well extend beyond the mixed layer in which phytoplankton populations develop, giving rise to a deep Chlorophyll *a* maximum. The modelled results do not account for these situations because the optical properties of the water column are modelled to be homogeneous with depth.

The sensitivity of Secchi depth to variations of CDOM at natural levels was observed to be highest in the Gulf of Finland and the Bothnian Bay (Table 2.1). In these areas, the response of Secchi depth at the target level was 0.5 m for both the summer and spring periods. The results suggest that the natural variations at the target level would be at least of this order of magnitude, when adding the possible minor sources of variations besides those derived from the light attenuation of CDOM. Of the other sub-basins, the Bothnian Sea and the Baltic Proper showed some deviations caused by

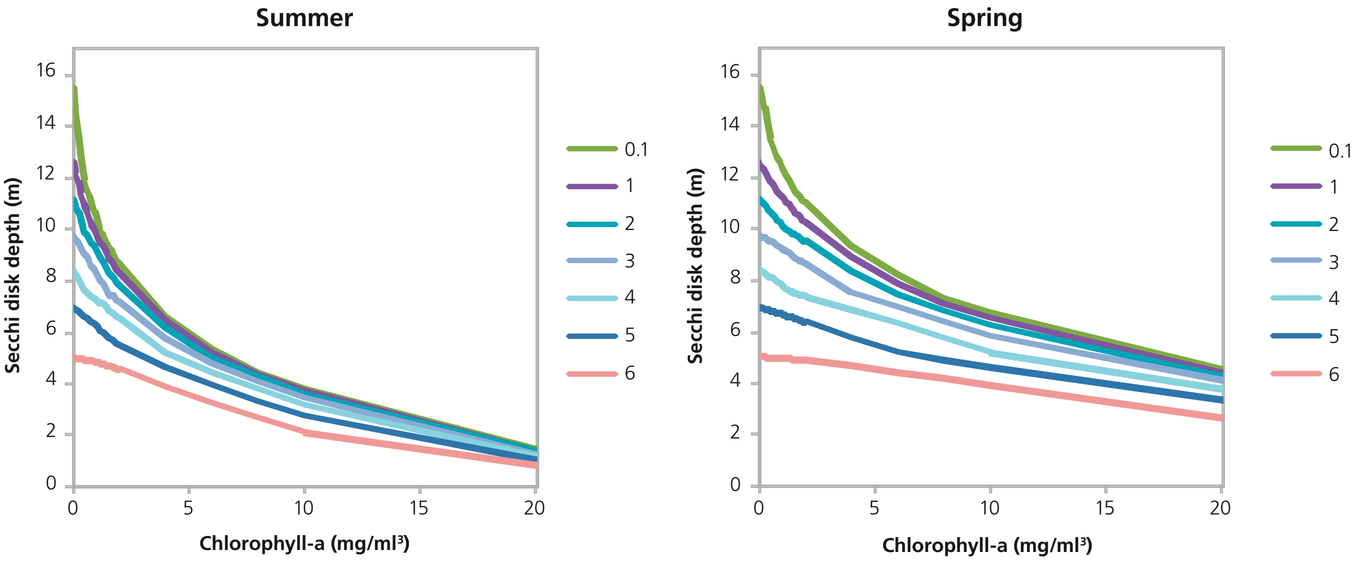


Figure 2.21 *Simulated Secchi depth as a function of Chlorophyll a concentration for seven simulated conditions of absorption by CDOM (m^{-1}), during summer and spring. These simulations are corrected by observations (see materials and methods, Fig. 2.20 for explanation).*

CDOM variations, while the southern sub-basins showed none (Table 2.1).

When comparing the long-term time series of the actual Chlorophyll *a* observations and Secchi-simulated values (Fig. 2.22), indications of a deviation going back in time might be observed in the Bothnian Bay and the Gulf of Finland. This deviation, though not proved by a long-time series nor a vast amount of data, can raise doubt to the assumption that CDOM absorption in these areas has remained constant in the long term. Moreover, the concentration of organic carbon in the rivers running into these basins has increased during this period (Pärn & Mander 2011, Räike et al. 2012), which might be seen as a cause for the increase of organic matter in the basins. A possible increase of CDOM absorption, in turn, supports the adjustments in Secchi depth targets presented for these sub-basins.
In addition, a tendency of the observed Chl *a* being lower than the Secchi-simulated values was observed in all sub-basins. This could be caused by several factors, including differences in observation time, insufficient information on CDOM concentrations or a slight bias in the modelled results. It must be pointed out that due to the Chlorophyll *a* observations not extending to the reference period, this information cannot be used as evidence or grounds for adjustments of targets, but merely as support for earlier analyses.

Table 2.1 *The average CDOM absorption and standard deviation at 375 nm, the Secchi depth target level achieved by data mining and the deviation in Secchi depth when CDOM concentration is at the 97.5-percentile for the Baltic open sea sub-basins. The latter estimates the expected additional deviation from a target level Secchi depth caused by the natural variation in CDOM concentration. The analysis is carried out separately for the summer and spring periods, and the average deviation was calculated from these. See materials and methods for a detailed explanation.*

Sea-area	ANNUAL AVERAGE			SUMMER PERIOD		SPRING PERIOD	
	Average (and std.dev.) of CDOM abs. (m^{-1})	Secchi depth target (m)	Deviation from target (m)	Secchi depth target (m)	Deviation from target (m)	Secchi depth target (m)	Deviation from target (m)
Danish Straits	0.8 (0.06)	6.3	0.0	5.9	0.0	7.4	0.0
Arkona Basin	0.9 (0.1)	7.4	0.0	6.6	0.0	8.0	0.0
Bornholm Basin	0.8 (0.04)	8.1	0.0	7.1	0.0	8.2	0.0
Baltic Proper	1.1 (0.1)	8.8	0.3	7.3	0.0	8.5	0.5
Bothnian Sea	1.2 (0.3)	6.9	0.2	6.9	0.1	6.5	0.2
Bothnian Bay	2.2 (0.5)	6.4	0.5	6.6	0.5	6.6	0.5
Gulf of Finland	2.2 (0.6)	5.4	0.5	5.4	0.5	4.9	0.5

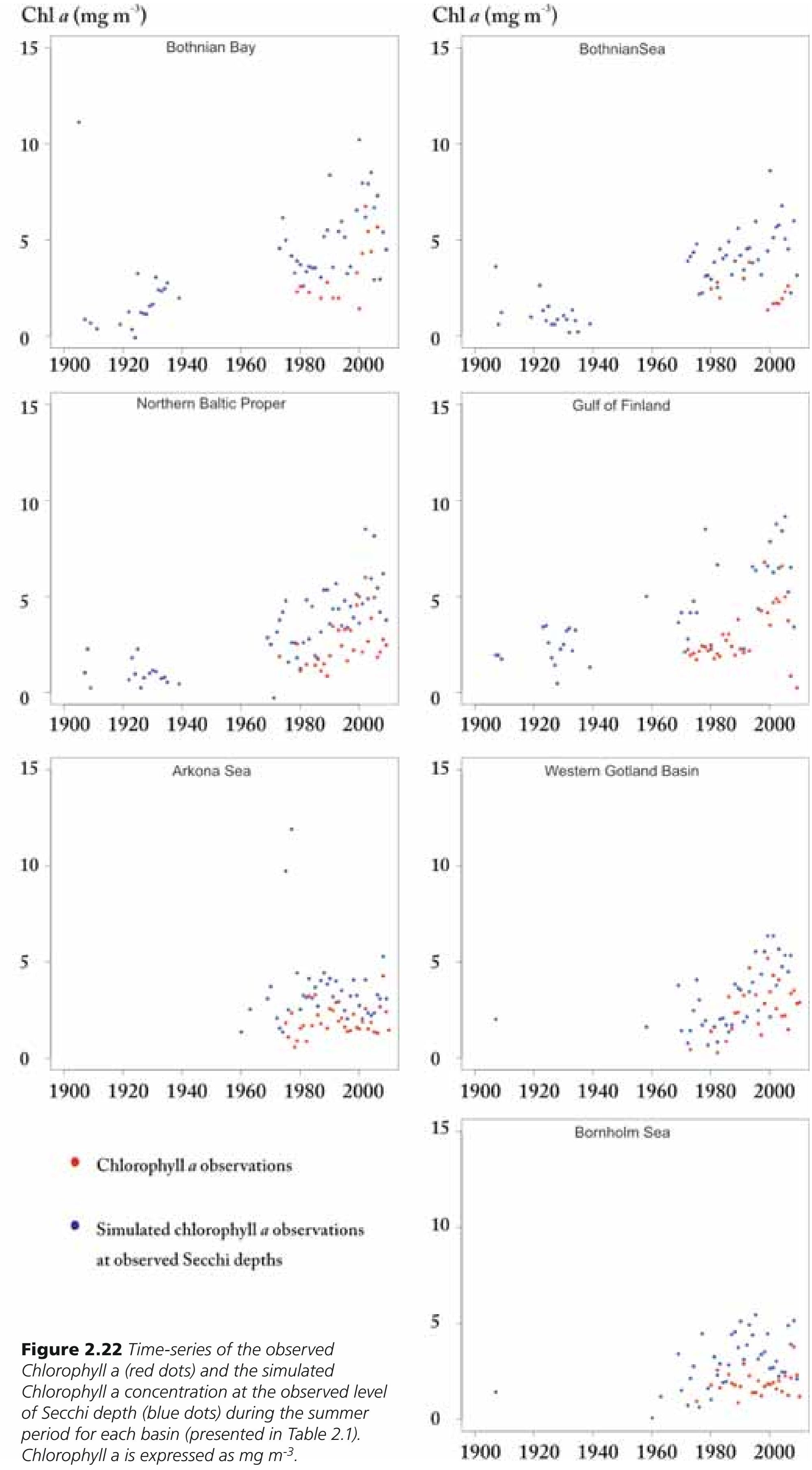


Figure 2.22 *Time-series of the observed Chlorophyll a (red dots) and the simulated Chlorophyll a concentration at the observed level of Secchi depth (blue dots) during the summer period for each basin (presented in Table 2.1). Chlorophyll a is expressed as mg m^{-3}.*

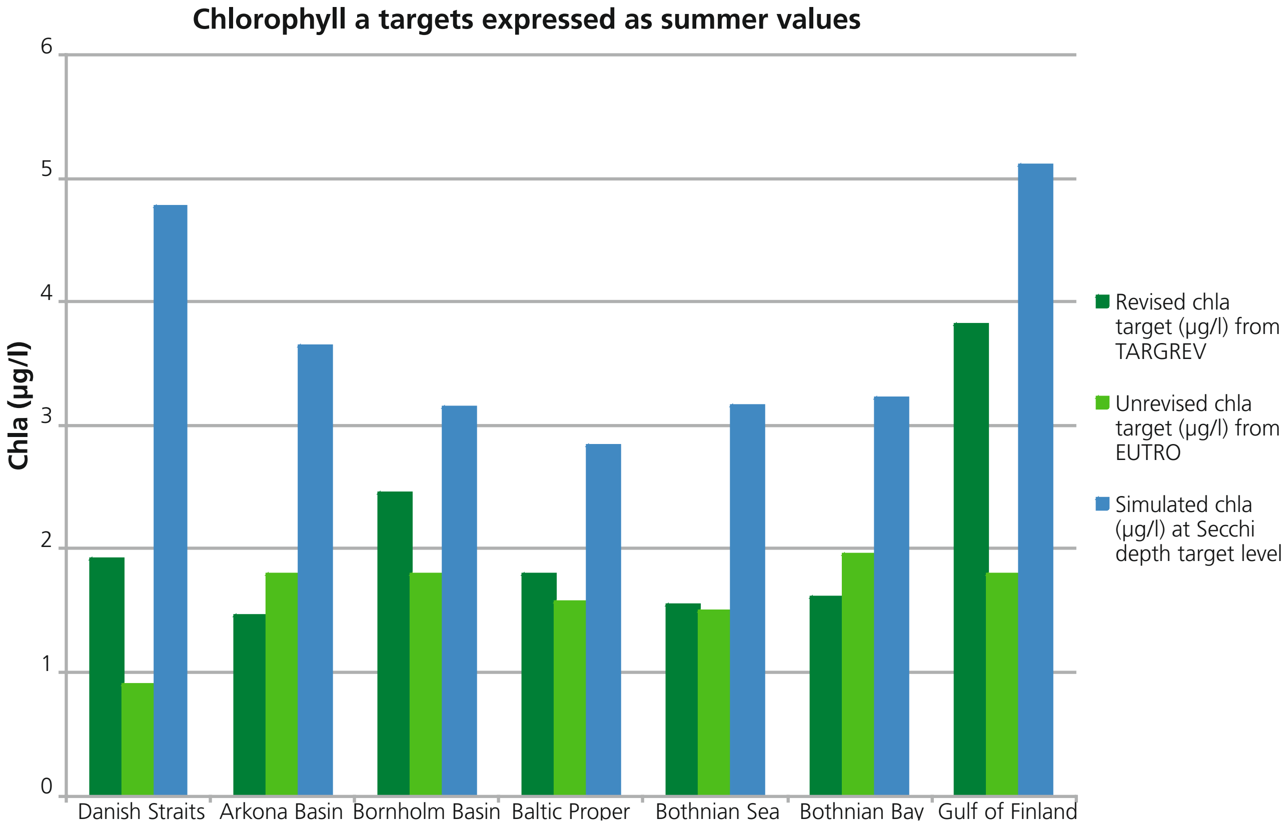


Figure 2.23 *Comparison of the Chlorophyll a targets (µg/l): the annual Chlorophyll a target found in the TARGREV data mining exercise, the earlier unrevised HELCOM EUTRO Chlorophyll a target, and the expected summer and spring period targets at the respective Secchi target levels. The three latter were transformed to an annual average in order to make a comparison. See materials and methods for further details.*

The bio-optically simulated Chlorophyll *a* concentrations at the Secchi depth target levels were observed to be somewhat different from the Chlorophyll *a* target concentrations found through data mining (Fig. 2.23). Based on the difference during the summer period, the simulated Chlorophyll *a* targets were, in comparison, 0.7 – 2.9 µg/l above the data mining targets. These results suggest that reaching Chlorophyll *a* targets would also naturally result in reaching the Secchi depth targets in the open Baltic sub-basins[5], assuming that the other optical properties of the water remain relatively constant.

The results and management suggestions made in this study are based on an optical model and restricted information on CDOM absorption in the Baltic sub-basins; the optical properties of changing phytoplankton communities has not been taken into account at a sub-basin level. In addition, the possibility of heterotrophic organisms affecting the attenuation of light is not examined due to a lack of data.

It can be summarized that based on the bio-optical studies:

1. The Secchi depth targets should be adjusted by +0.5 m in the Bothnian Bay and the Gulf of Finland to account for possible changes in CDOM absorption.
2. The suggested Chlorophyll *a* targets are sufficient in order for the suggested Secchi depth targets to be reached, assuming that the other optical properties of the water remain relatively constant.

5 Bearing in mind that the Gulf of Riga was not included in the study, due to unsufficient data.

2.4 Oxygen conditions

In the HELCOM system of Ecological Objectives (EcoOs), oxygen conditions are directly linked to the EcoO “natural oxygen levels”. Oxygen is also one of the HELCOM BSAP indicators for eutrophication. It has subsequently been used as one of the core indicators of eutrophication in the HELCOM integrated thematic assessment of eutrophication (HELCOM 2009) and the HELCOM Initial Holistic Assessment of the Ecosystem Health of the Baltic Sea (HELCOM 2010). In addition, it is one of the indicators of eutrophication describing good environmental status (GES Descriptor 5 and supporting Descriptor 6) in the Marine Strategy Framework Directive, as described in Commission Decision 2010/477/EU.

Hypoxia (defined as oxygen concentrations below 2 mg l^{-1}) is one of the most deleterious effects of eutrophication, which has increased substantially over the last century (Johnsson et al. 1990, Österblom et al. 2007, Zillén & Conley 2010). Lamination indicates that hypoxic conditions in the sediments have occurred at the deepest depths during the Holocene in response to climatic variability and potentially also anthropogenic activity (Karlson et al. 2002, Zillén & Conley 2010); however, these studies bear no evidence of hypoxia in the water column. Nonetheless, the present extent of hypoxia, both in the water column and sediments, is unprecedented and the largest change has occurred since the 1950s (Conley et al. 2009).

Bottom water oxygen concentrations are strongly influenced by physical factors, especially the inflow of saltier, denser water. These inflows are governed by large-scale and local meteorological forcing, and have large variations in frequency and magnitude over time-scales of decades (Meier et al. 2006, Meier 2007, Lass & Matthäus 2008, Reissmann et al. 2009). Salt water inflows may supply oxygen to bottom waters, but at the same time they enhance stratification with the potential of expanding bottom areas at risk of experiencing hypoxia (Conley et al. 2002, 2009). Thus, salt water inflows may improve oxygen conditions in bottom waters in the short term, but in the longer term they reduce the vertical mixing of oxygen. Salt water inflows are a natural process perturbing the bottom waters and hence the oxygen conditions in the bottom waters. However, it is the increased flux of organic material to the bottom water and sediments due to nutrient enrichment that has disrupted the subtle balance between the oxygen supply and oxygen consumption from the decomposition of organic material. Since oxygen consumption below the halocline (deep water layer) is linked to the amount of particulate organic material sedimenting from the surface layer, the cumulative lack of oxygen (oxygen debt) is an integrative indicator of the state of eutrophication; however, perturbations from the physical forcing of the system overlay this anthropogenic trend.

Although oxygen conditions have undoubtedly worsened over the last century, as evidenced by sporadic observations from before the late 1960s when more consistent monitoring was initiated, unbiased quantitative estimates of the trends in oxygen concentrations have not been reported yet. There are a number of problems comparing earlier data with the newer, more consistent monitoring data that have to be resolved in order to produce consistent estimates of trends in oxygen conditions:

1. Observations are sparsely distributed in both time and space.
2. The entire water column is not always monitored; in particular, profiles in earlier data do not always reach the bottom.
3. Profiles in earlier data are characterised by a few discrete samples.
4. Oxygen measured with the CTD sensor occasionally have an offset relative to water samples measured by the Winkler titration.
5. Hydrogensulphide (H_2S) has been measured since 1960 but not consistently. When measured, H_2S has been converted into a negative oxygen concentration to describe the amount of oxygen needed for oxidation before measurable oxygen concentrations can be expected. The oxidation potential for other substances such as NH_4, Fe and Mg are not included. Thus, an oxygen concentration equal to zero may actually represents conditions with H_2S and therefore a negative oxygen concentration. Such observations that actually represent values below their nominal values are termed censored data.

In this section, statistical methods to analyse oxygen data - given the associated problems above - are developed to provide unbiased and consistent estimates for the trends in Baltic Sea hypoxia.

The objective of this analysis was to estimate the volume and the extent of hypoxia over time, as well as the total oxygen debt, defined as the "missing" oxygen relative to a full saturated water column.

2.4.1 Methods

The heterogeneity of the profiles across the spatial domain - sampled as part of regular monitoring programs and research projects - was resolved by aggregating the locations of all profiles on the existing network of HELCOM/National monitoring stations (Fig. 2.24). This resulted in 339 monitoring stations for the Baltic Sea behind the sills at Drogden and Darss. The statistical analysis described below was not carried out for the Kattegat and the Danish straits since hypoxia in this area is seasonal and somewhat dynamic, requiring good data coverage to estimate the extent and volume of hypoxia. Thus, estimating the extent of hypoxia for this area is only achievable for recent years.

Profile estimations

The sparseness and resolution of oxygen profiles in the earlier data naturally impose bounds on the complexity of the statistical analyses of such data. As the oxygen profiles in the earlier data typically have less than ten discrete samples, it is not possible to interpolate these data with depth because of the poor resolution, particularly around the halocline where the oxygen concentrations change drastically. The aim here is to model the oxygen profile using information from the salinity profiles, which typically has a higher vertical resolution than the oxygen in the older data. From the more recent profiles, and partly also the older profiles, it is observed that: 1) oxygen concentrations can be supersaturated in the euphotic zone during the phytoplankton growth season; 2) oxygen concentrations are close to saturation to about the start of the halocline; 3) oxygen concentrations decline rapidly with depth across the halocline; and 4) oxygen concentrations may increase or decrease with depth below the halocline. Thus, the assump-

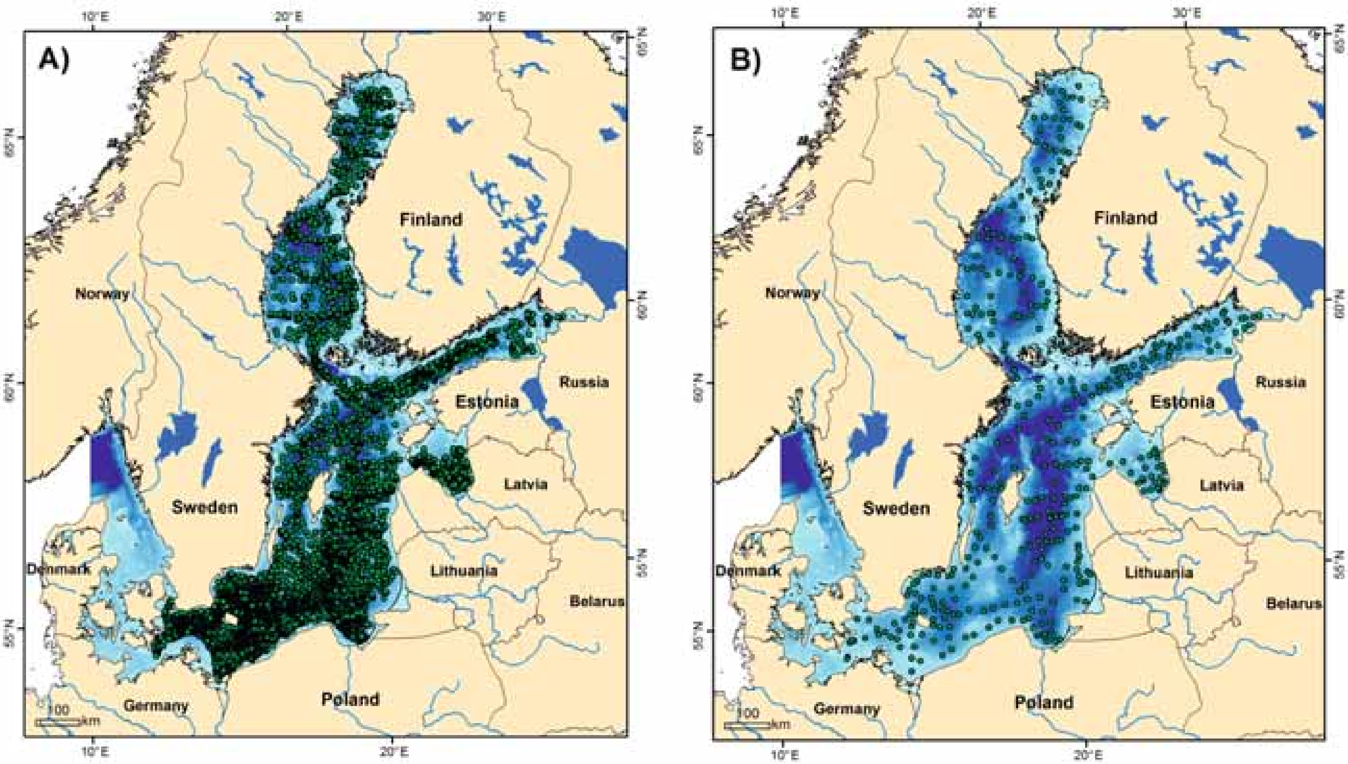


Figure 2.24 *A) Positions of all profiles used in the analysis of the oxygen objective. B) Aggregation of the profiles onto the existing monitoring network.*

tions underlying this typical oxygen profile are that the sharp decline in oxygen concentrations across the halocline is due to low mixing and, to some extent, the decomposition of trapped organic material from the surface layer.

Horizontal inputs of oxygen with saltwater inflows may improve oxygen conditions below the halocline, resulting in small changes in oxygen below the halocline whereas the lack of horizontal ventilation of the deeper layers will lead to a stronger decline in oxygen concentrations with depth, as typically observed during periods with few saltwater inflows.

The estimation of such simple oxygen profiles follows a two-step procedure (Fig. 2.25), where the salinity profile is modelled in the first step and used to identify the depths defining the halocline (change points in Fig. 2.25B); in the second step, these depths (change points) are used for defining the linear segments of the oxygen model. The salinity profile is typically sigmoid-shaped around the halocline depth, which was modelled using the cumulative normal distribution function to yield a symmetric salinity profile around the halocline.

The cumulative normal distribution function was fitted to each monitored salinity profile using the average salinity between 20 and 30 m (Bornholm Basin and the Baltic Proper) or between 10 and 20 m (other basins) (hence avoiding potential freshwater plumes on the very surface) as the base for the distribution and fitting three parameters describing:

1. The salinity difference across the halocline (scaling factor for distribution, i.e. the cumulative distribution functions range from 0 to 1).
2. The halocline depth (mean of distribution).
3. The change in salinity with depth (±standard error of distribution) or steepness of the halocline. The largest change (~70%) in salinity occurs within one standard error range of the halocline; this interval was used to define an upper and lower change point for the density profile.

The two change-points describe the layer of the halocline, defined as the permanent discontinuity layer in Lass & Matthäus (2008). Examples of fitting this sigmoid function to the salinity profile will be given in the results.

Although this density model is rather simple - it only includes three parameters - it is not guaranteed that the estimation of these parameters will always result in a meaningful description of the salinity profile. In particular, unreliable parameter estimates can be obtained if the measured profile only spans part of the expected sigmoid model. Therefore, the parameter estimation was constrained to profiles having a depth that gen-

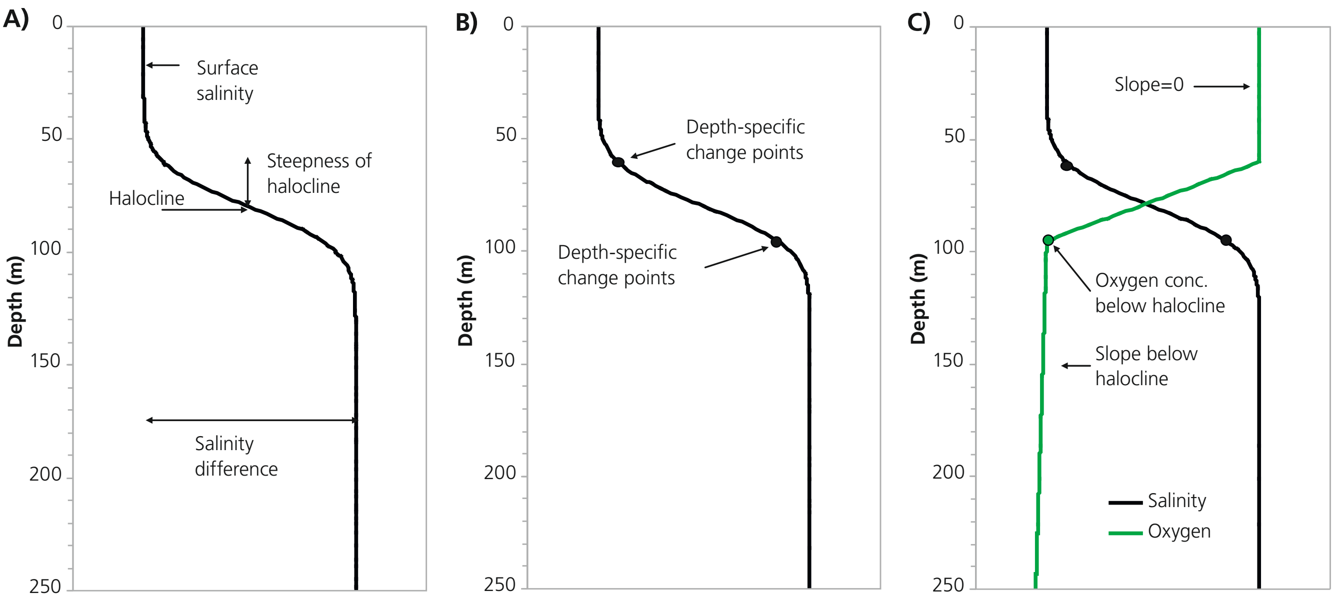


Fig. 2.25 *The procedure for estimating a linear segmented oxygen profile. A) Sigmoid representation of the density profile (cumulative functions of the normal distribution). B) Identification of change-points in the density profile. C) Estimation of the segmented linear model for oxygen using the identified change-points.*

erally included large parts of the halocline, i.e. exceeding 80 m for the Baltic Proper and Gulf of Finland; exceeding 60 m for the Bornholm Basin and the Bothnian Sea; and exceeding 40 m for the Arkona Basin, Gulf of Riga and Bothnian Bay. This implies that profiles from the rim of the deep water basins were discarded and are thus not relevant for characterising oxygen conditions below the halocline. Secondly, the convergence of the estimation algorithms was not always achieved since the sigmoid relationships were fit by non-linear regression methods with relatively wide bounds on the parameter space. Profile parameter estimates from algorithms that did not converge or ended on the bounds on the parameter space were discarded.

Oxygen concentrations vary across the Baltic Sea because of gradients in salinity and temperature that affect the oxygen saturation as well as in response to eutrophication. In reality, oxygen saturation can vary from around 9 mg l^{-1} during summer in the Arkona Basin to over 14 mg l^{-1} at low temperatures (~0°C) and salinity in the Bothnian Bay. Therefore, in order to extract a signal that excluded variations due to the solubility of oxygen in seawater, oxygen concentrations were converted into oxygen debts by subtracting the measured concentration from the saturated concentration, calculated from the salinity and temperature of the given parcel of water.

Having defined the two change points from the salinity profile (Fig. 2.25B), two parameters were used to characterise the oxygen debt profile. One parameter described the oxygen debt in the lower change point, i.e. the oxygen concentration below the halocline; the second described the change of oxygen debt with depth below the change point (cf. Fig. 2.25C). The discrete oxygen profile was interpolated with depth, using salinity as the covariate for the interpolation to estimate the oxygen concentration in the lower change point; the second parameter was found as the average change in oxygen debt with depth from this point. Since the oxygen profile below the halocline may contain censored data (i.e. oxygen concentrations ~0 and no H_2S measurement taken), a censored-data regression approach was used (see e.g. Carstensen (2010) for details).

Trends in profile parameters

The profile parameters were partitioned into six basins (Arkona Basin, Bornholm Basin and the Baltic Proper including the Gulf of Finland, Gulf of Riga, Bothnian Sea, and Bothnian Bay). For each basin, a general linear model (GLM) was first fitted for each of the salinity and oxygen profile parameters using three factors (station, month and year), i.e. the model partitioned variations in the profile parameters into spatial variation, seasonal variation and trend (see Carstensen et al. 2006 for details). Based on the station-specific mean estimates of the profile parameters, a two-dimensional spline was fit to derive a smooth spatial gradient (GAM model) covering the entire study area. This smooth spatial model was used to spatially detrend the data by subtracting the spatial model predictions for the different parameter estimates.

After the spatial detrending all profile parameters were analysed with a simpler GLM model that only described the trend and seasonal variation, it was estimated for each basin separately to allow basin-specific seasonality and trends. A number of profiles still had parameter estimates that deviated substantially from the overall pattern of the profile parameters. Therefore, a robust regression model was employed for the five parameters (three parameters for the salinity profiles: salinity difference, position and steepness of the halocline; and two parameters for the oxygen profiles: oxygen debt below the halocline and loss of oxygen with depth below the halocline) describing the variation between years and months, and outliers (profile parameter estimates outside of the 99.9% confidence prediction interval of the model) were discarded and the regression model re-estimated. The parameters were replaced by the regression model predictions for those salinity profiles where the estimation did not converge or the parameter estimates were considered outliers according to the robust regression algorithm (approximately 5% of all profiles).

Integration of salinity and oxygen over basins

The procedures above resulted in a smooth spatial trend, a seasonal pattern (given by 12 monthly values for each basin) and a long-term trend for each of the five profile parameters. The spatial model and the long-term trend described the variations as an average over all 12 months. The spatial model and long-term trend values were used for each basin separately to horizontally and vertically integrate the volume below the halocline; the total amounts of salinity above and below the halocline; the area and volume of hypoxia (<2 mg l^{-1}); and the total oxygen debt below the halocline (i.e. from the lower change-point in Fig. 2.25 and below).

Linking oxygen trends to nutrient inputs and physical forcing

The oxygen debt is strongly linked to the volume of bottom water below the halocline, since the oxygen debt is the product of the volume and an average oxygen debt concentration below the halocline (i.e. oxygen debt = $V_{bottom} \times C_{O2,bottom}$). Therefore, let us consider the oxygen debt in a parcel of water below the halocline, and let us first investigate this volume-specific annual oxygen debt under quasi-steady state assumption between oxygen loss and supply, acknowledging that is not the case in the Baltic Sea.

Oxygen loss depends on the amount of sedimenting organic material, which is assumed proportional to the volume-specific nutrient input (N_{input}/V_{bottom}) in addition to a base respiration rate (a_0). The base respiration accounts for oxygen consumption from other processes that are not strictly correlated to the nutrient input. This implies that the organic loading of the bottom water will be relatively larger when the volume of the bottom water is small, given that particulate matter produced in the surface layer will sediment below the halocline and consume the same amount of oxygen, whether the bottom volume is small or large.

Oxygen is supplied to the bottom water through horizontal transport and vertical mixing, although both of these involve several different mechanisms (Meier et al. 2006, Reissmann et al. 2009). For the horizontal transport, two sources are considered: 1) Major Baltic Inflows (MBI), as defined in Fischer & Matthäus (1996) and Matthäus et al. (2008), assumed to replenish the deep bottom waters with saline oxygen-rich waters; and 2) ordinary horizontal transport proxied by variations in the bottom water salinity (S_{bottom}). For the vertical mixing, the Brunt-Väisälä frequency (N_{BV}) was calculated (vertical mixing is proportional to $1/N_{BV}$) from the estimated salinity difference across the halocline and the halocline steepness. The vertical mixing of oxygen is therefore the concentration difference, given by the volume-specific oxygen debt ($C_{O2,bottom}$, since $C_{O2,surface}$=0), divided by the Brunt-Väisälä frequency. Thus, equating oxygen loss and supply on an annual scale yields

$$a_0 + a_N \times N_{input} / V_{bottom} = a_{MBI} \times MBI + a_{salinity} \times S_{bottom} + a_{BV} \times CO2_{,bottom} / N_{BV} \qquad \text{(Eq. 2.1)}$$

where a_i's are scaling parameters for the different fluxes. Reorganising this equation to isolate the volume-specific oxygen debt gives

$$C_{O2,bottom} = (a_0 + a_N \times N_{input}/V_{bottom} - a^{MBI} \times MBI + a_{salinity} \times Sbottom) \times N_{BV}/a_{BV} \qquad \text{(Eq. .2)}$$

Thus, the volume-specific oxygen debt in the bottom water is a balance between oxygen loss and horizontal transport, modulated by the mixing across the halocline. Equation (2.2) constitutes a non-linear regression model, albeit it is over-parameterised in the sense that there are four parameters but only three independent terms on the right-hand side of equation (2.2). Thus, fixing a_{BV}=1 implies that the other parameters (a_N, a_{MBI}, and $a_{salinity}$) are estimated relative to the "true" and unknown value of a_{BV}. Equation (2.2) was estimated using annual values for N_{input} (nitrogen and phosphorus separately), MBI and S_{bottom} from the same year and the year before to describe a potential lagged response. Moreover, annual values of N_{input} and MBI were calculated using the period October-September to account a delay in the oxygen response: 1) Nutrient inputs after September are unlikely to stimulate primary production that same year and will accumulate for the following productive season; and 2) the horizontal transport of MBI to the Bornholm Basin and Gotland Deep is expected to be around 2-4 months (Feistel et al. 2003), with MBI generally occurring during the winter period. The potential cumulative effect in the volume-specific oxygen debt (Eq. 2.2) was accounted for by adding an autoregressive term to Equation (2.2), such that the volume-specific oxygen debt in one year would depend on the oxygen debt of the previous year.

The regression model (Eq. 2.2) was estimated for the Bornholm Basin and Baltic Proper only, since this is where hypoxia is perennial. Input variables lagged one year on the right-hand side of Equation

(2.2) were not significant and were removed and the model re-estimated. The two region-specific models were estimated using both nitrogen and phosphorus for nutrient inputs. Temporal variations in $C_{O2,bottom}$ was partitioned into a climate trend by subtracting the estimated component relating to nutrient input and an anthropogenic trend by subtracting the two components pertaining to MBI and S_{bottom}. Residuals from the models were examined for trends to identify a potential change over time in the estimated relationship.

2.4.2 Results and discussion

Results from the spatial models will be shown for the entire study area (east of the Drogden and Darss sills), whereas time trends are shown for the Baltic Proper and the Gulf of Finland that were combined (denoted as Baltic Proper) into a single basin for the modelling. For the description of hypoxia, the Gulf of Finland can be considered an extension of the Baltic Proper, since there is no sill dividing the two waters. There is not a permanent halocline in the Gulf of Riga and therefore only the surface salinity in this basin is considered. Time trends from other basins can be found in Annex C together with seasonal variations in the profile parameters for all basins.

Salinity profiles

A total of 39,933 salinity profiles were parameterised by fitting a sigmoid function; additionally, 91,889 surface salinities were calculated from open water stations, showing that less than half of the salinity profiles were deep enough to characterise the halocline. The salinity profile generally fitted quite well to the observed data, although there could be deeper intrusions not covered by the model (Fig. 2.26). The observed oxygen profiles were not as systematically shaped with depth as the salinity due to multiple layers having different oxygen levels, particularly after intrusions. Due to vertical diffusion and mixing, the oxygen profile will inherently aim at stabilising with a gradual decline in oxygen concentrations; however, salt water intrusions with different oxygen levels at various depths will maintain irregularities in the oxygen profile (exemplified in Fig. 2.26C). The parameterised oxygen profiles, however, provided a robust estimate of the average oxygen concentration below the halocline. Above the halocline, the parameterised oxygen profile generally did not describe the observed oxygen concentrations well, but this is of less concern since this part of the water column will be mixed with the surface layer when the thermal stratification is broken down.

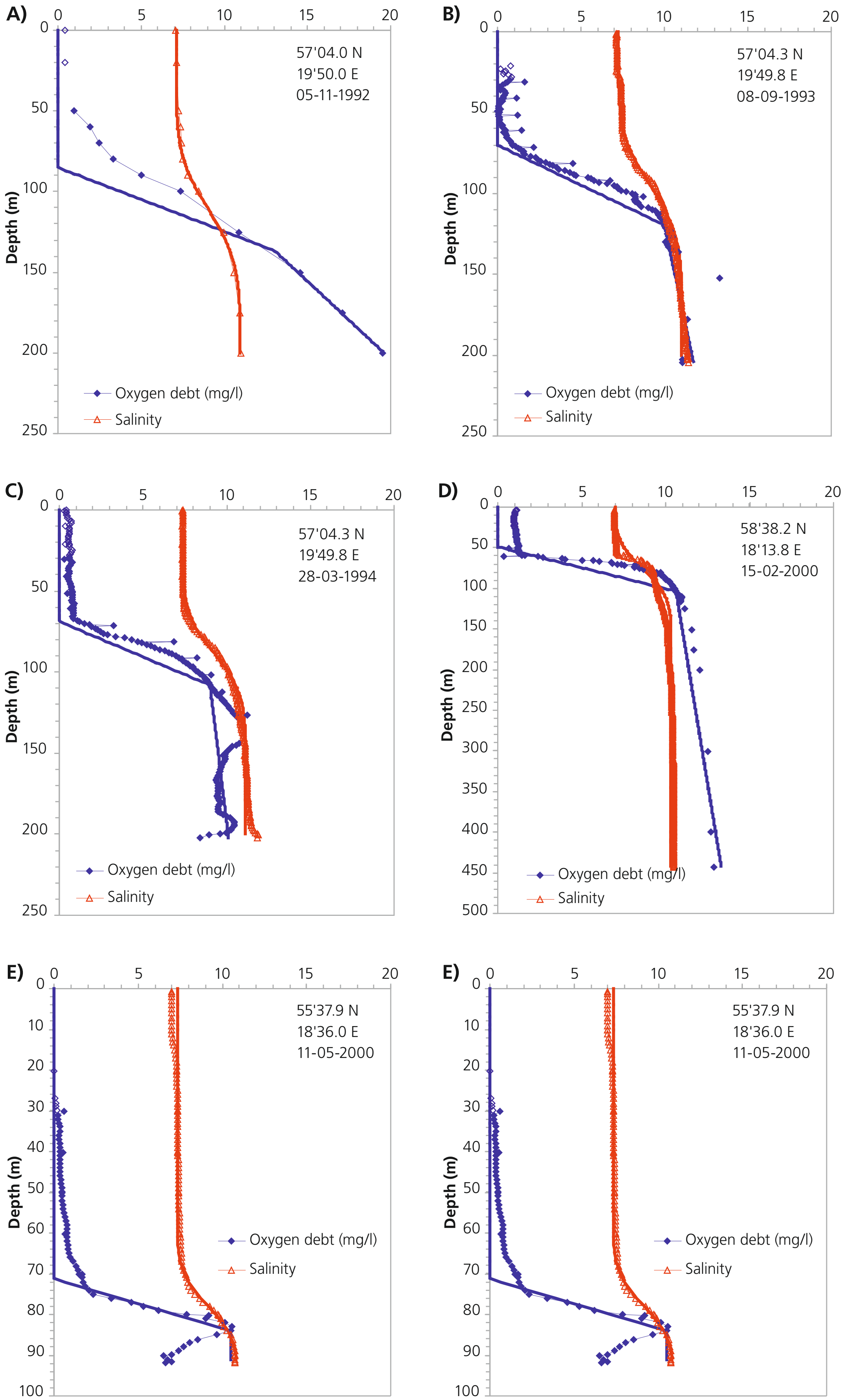


Fig. 2.26 *Examples of parameterisation of salinity and oxygen debt profiles from six profiles in the Baltic Proper. The oxygen debt exceeds the oxygen saturation when H_2S is present. The observations are shows as points and fitted profiles by solid lines.*

The spatial distribution of sub-surface (20-30 m) salinity (Fig. 2.27A) and salinity difference between the sub-surface and the bottom (Fig. 2.27B) showed the expected pattern with a decreasing sub-surface salinity gradient from southwest to northeast, which was also partially reflected in the salinity difference, albeit with a depth dimension added to it so that deeper areas tended to display

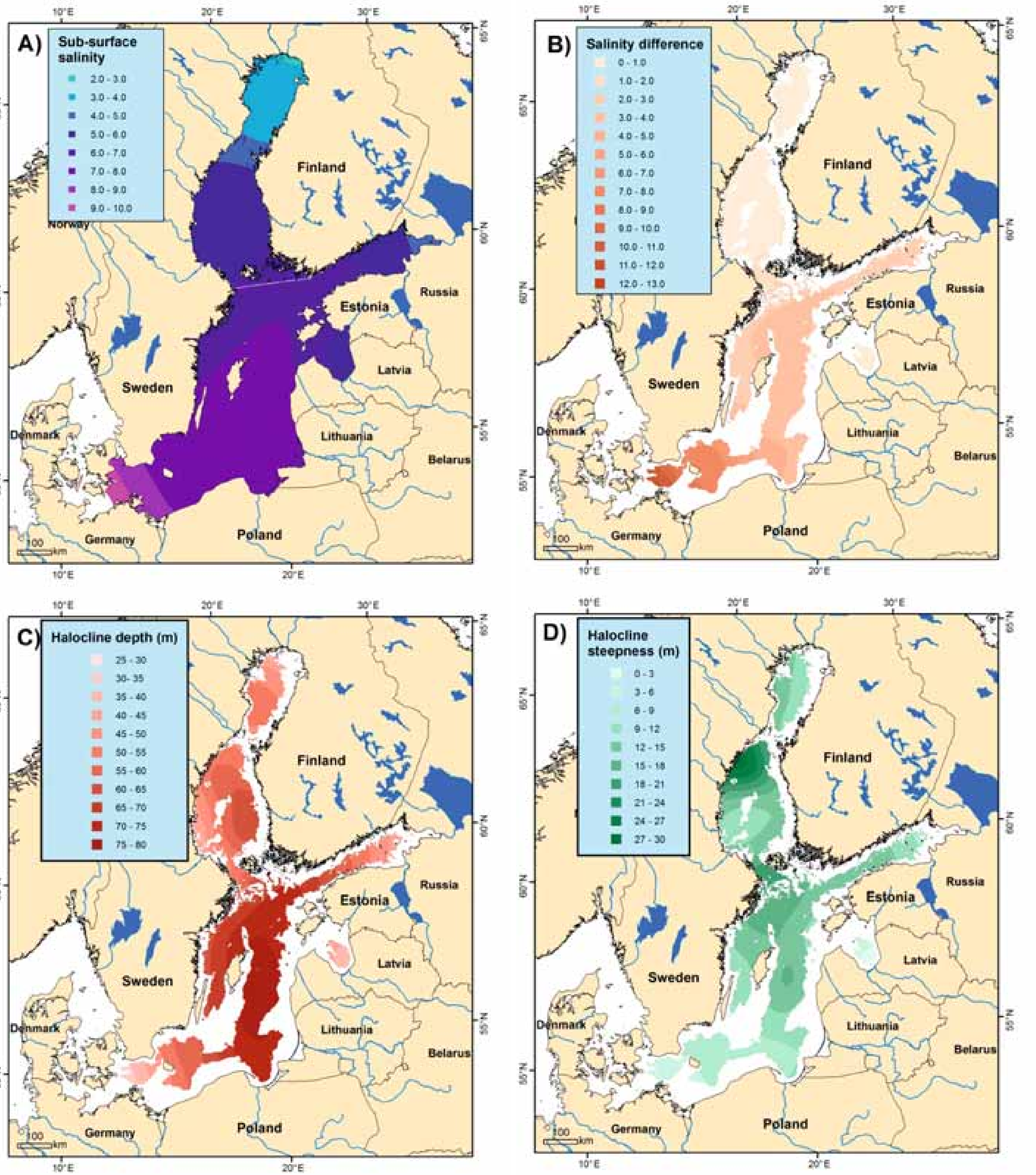


Figure 2.27 *Spatial distribution of A) sub-surface salinity (20-30 m); B) salinity difference between the sub-surface and the bottom; C) halocline depth; and D) range of halocline. The spatial distributions are means of the period 1900-2010. In B), C) and D), the areas are coloured for water depths below the average depth of the halocline, i.e. salinity stratified areas only.*

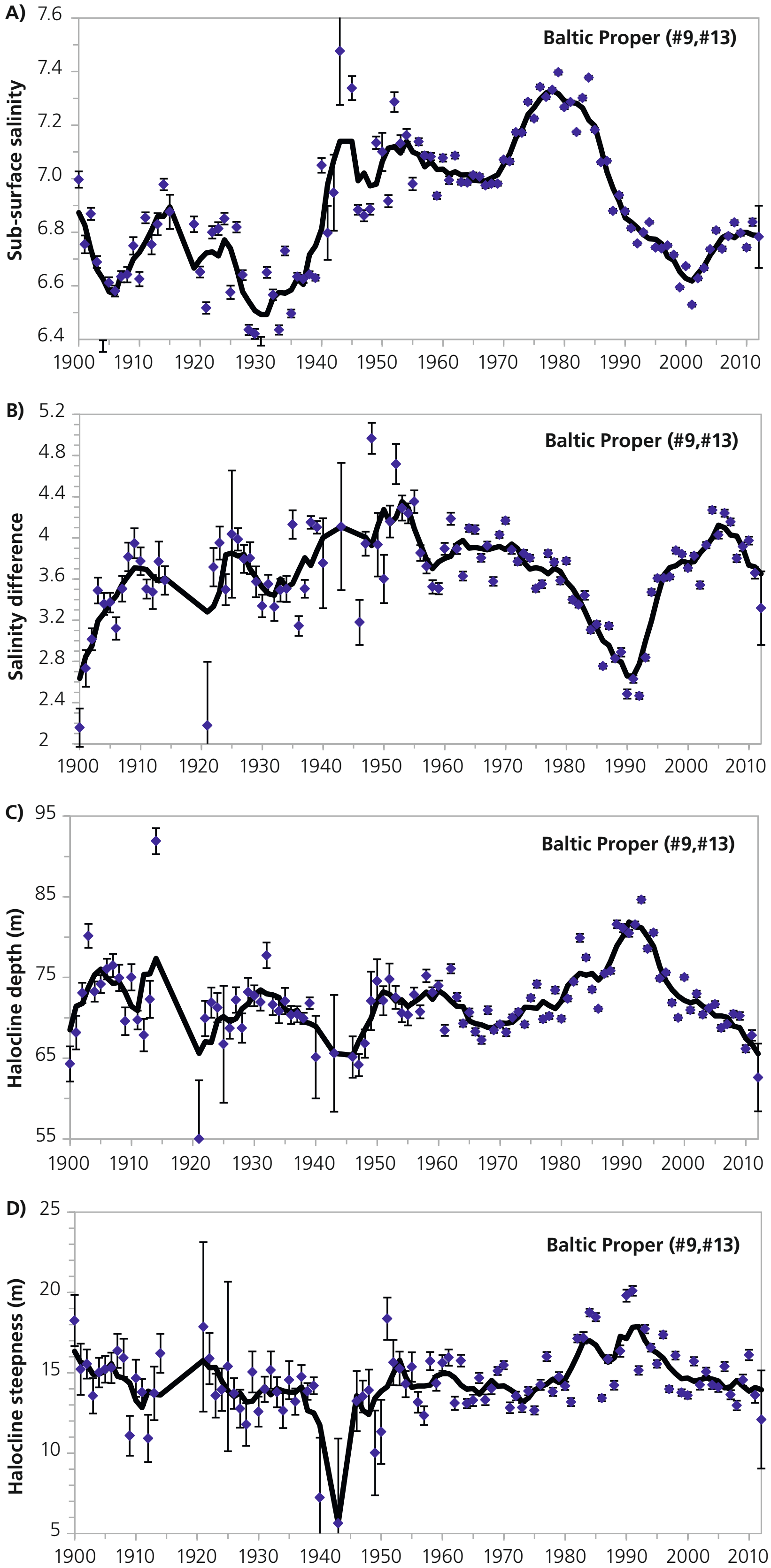


Figure 2.28 *Trends in A) sub-surface (20-30 m) salinity; B) salinity difference between sub-surface and bottom waters; C) depth of halocline; and D) halocline steepness estimated as the standard error in the profile model. The solid line is the five-year moving average.*

a larger difference. The depth position of the halocline was around 35-40 m in the Arkona Basin increasing to about 60 m in the Bornholm Basin and deepening even further in the Baltic Proper to about 70-80 m (Fig. 2.27C). In the Gulf of Finland, the location of the halocline went up going east, ending at around 45 m in the eastern-most salinity stratified waters. Similarly, the depth of the halocline in the Bothnian Sea was around 50-70 m moving slightly up to 45-60 m in the Bothnian Bay. The steepness of the halocline (Fig. 2.27D) was around 5 m in the Arkona Basin increasing to about 8 m in the Bornholm Basin and even further to 15-20 m in the Baltic Proper. Furthermore, there was no sharp halocline between the Northern Baltic Proper and the Bothnian Sea; eroded haloclines were also observed, even more pronounced, in the Northern Bothnian Sea. In the Bothnian Bay, the halocline steepness was typically 15 m.

The sub-surface salinity in the Baltic Proper has varied by almost one unit within the last 110 years (Fig. 2.28A). Sub-surface salinity decreased from 6.8 to about 6.5 in the 1930s and then increased around 1940 where the salinity oscillated between 7 and 7.4 until the mid-1980s when it dropped to the more recent level of around 6.7. Particularly pronounced is the drop in salinity from 7.4 to 6.7 during the 1980s and 1990s. This period from 1983 to 1993 (termed the stagnation period) was characterised by low frequency of saltwater intrusions according to the index proposed by Matthäus & Franck (1992). This period was also apparent in the trend of the salinity difference (Fig. 2.28B) where there was a salinity drop of more than one unit. These variations suggest a strong freshening of the Baltic Sea during the stagnation period, but it is also apparent that the Baltic Sea was fresher during the first half of the 20th century, similar to what was reported in Fonselius & Valderama (2003). Despite the low salinity up to around 1940, the halocline was rather steady at around 70-75 m, whereas the stagnation period deepened the halocline by almost 10 m to a mean of about 80 m (Fig. 2.28C). However, after the major intrusion in 1993, the halocline depth has returned to the more general level of 70-75 m. Finally, the halocline steepness has varied around 15 m (Fig. 2.28D) with some low values in the 1940s that could be associated with saltwater inflow and strengthening of the halocline (cf. Fig. 2.28A); however, these are based on very few profiles. There was also a tendency to a more eroded halocline, indicated by an increase in the halocline steepness, during the stagnation period (Fig. 2.28D). In general, there have been changing salinity levels in the Baltic Sea; for analysing hypoxia however, the most important characteristic is the halocline distribution and range. Throughout the 110-year study period, and with the exception of the stagnation period, the halocline has been rather stationary with a depth of around 70-75 m and a steepness of about 15 m. This corresponds to the observation that the major of the salinity change is between 55 and 90 m.

Oxygen profiles

The spatial distribution of the oxygen debt below the halocline (Fig. 2.29A) and loss of oxygen with depth (Fig. 2.29B) showed that oxygen concentrations below the halocline in the Bothnian Sea and Bothnian Bay were close to saturation (oxygen debt close to zero) over the study period (1900-2010). Oxygen debts below the halocline were also low in the Arkona and Bornholm Basins, as well as the Eastern Gulf of Finland. However, oxygen debts were high (8-10 mg l^{-1}) in large parts of the Baltic Proper, corresponding to hypoxic conditions (<2 mg l^{-1}) starting just below the halocline. On the other hand, the loss of oxygen with depth was lower in these areas of the Baltic Proper (0.01-0.05 mg l^{-1} m^{-1}), whereas oxygen concentrations declined fast with depth in the Arkona and Bornholm Basins, as well as in the Eastern Gulf of Finland (Fig. 2.29B). The loss of oxygen with depth was relatively low in the Bothnian Sea (0.01-0.05 mg l^{-1} m^{-1}) and even lower in the Bothnian Bay (0.01-0.02 mg l^{-1} m^{-1}). The loss of oxygen with depth in the different basins was largely related to the thickness of the bottom water layer.

The trends of the oxygen profile parameters both showed increasing tendencies (Fig. 3.4.7), albeit most pronounced for the oxygen debt below the halocline. The oxygen debt just below the halocline was typically around 7-8 mg l^{-1} in the first half of the 20th century, but then increased up to ca. 1970 when it reached a level of around 10 mg l^{-1}. Oxygen conditions just below the halocline tended to improve during the stagnation period (1983-1993), reaching levels below 9 mg l^{-1}; following this, the oxygen debt increased and has been above 10 mg l^{-1} for the last decade. The loss rate of oxygen below the halocline was similarly low in

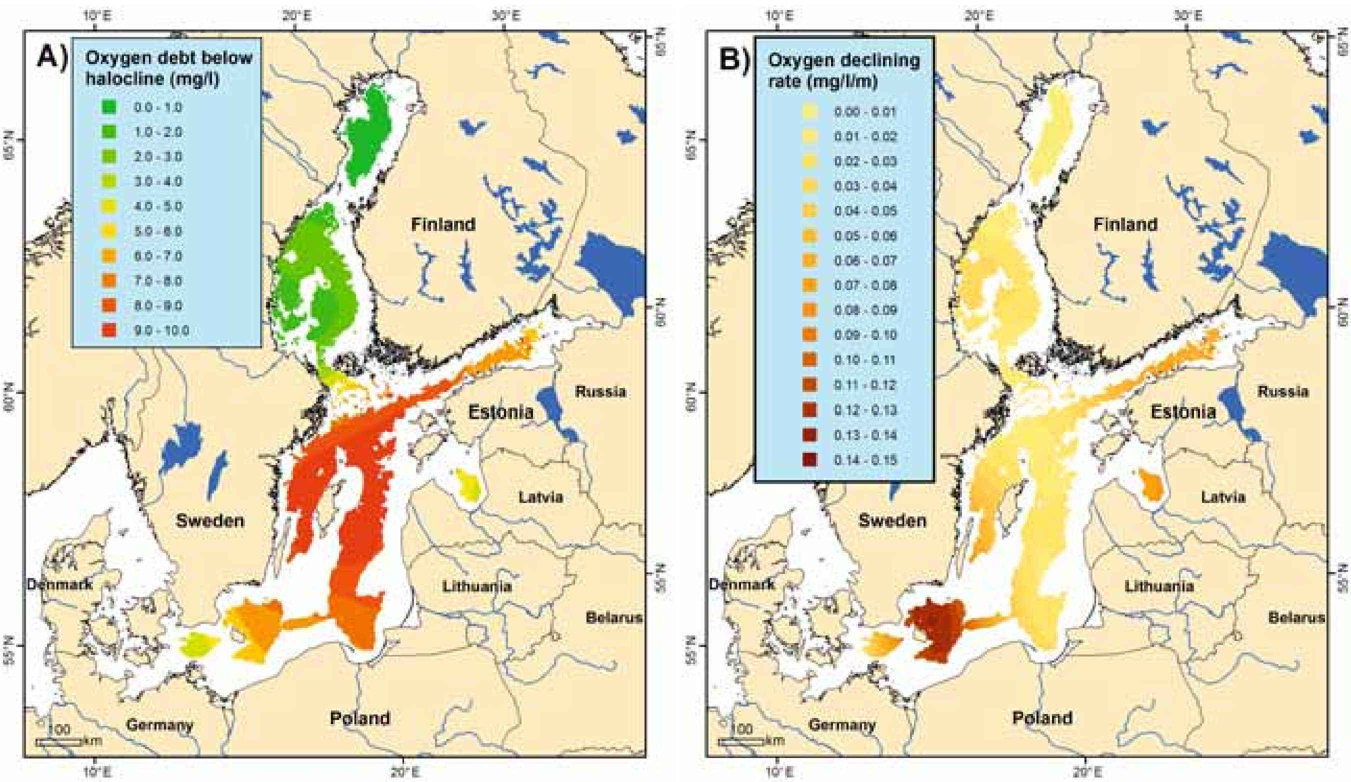


Figure 2.29 *Mean spatial distribution of A) oxygen debt just below the halocline; and B) loss rate of oxygen with depth over the entire study period (1900-2010).*

the first half of the 20th century; however, the estimates were also uncertain. Oxygen loss rates with depth continued to be low after 1950, perhaps with a slight decrease, until the start of the stagnation period when loss rates more than doubled to a level of 0.05 mg l^{-1} m^{-1}. After the large saltwater inflow in 1993, the loss of oxygen with depth has reached an intermediate plateau around 0.03-0.04 mg l^{-1} m^{-1}. Major saltwater inflows affect both parameters by increasing oxygen debt below the halocline and decreasing the loss rate of oxygen with depth, resulting in a more even depth distribution of oxygen.

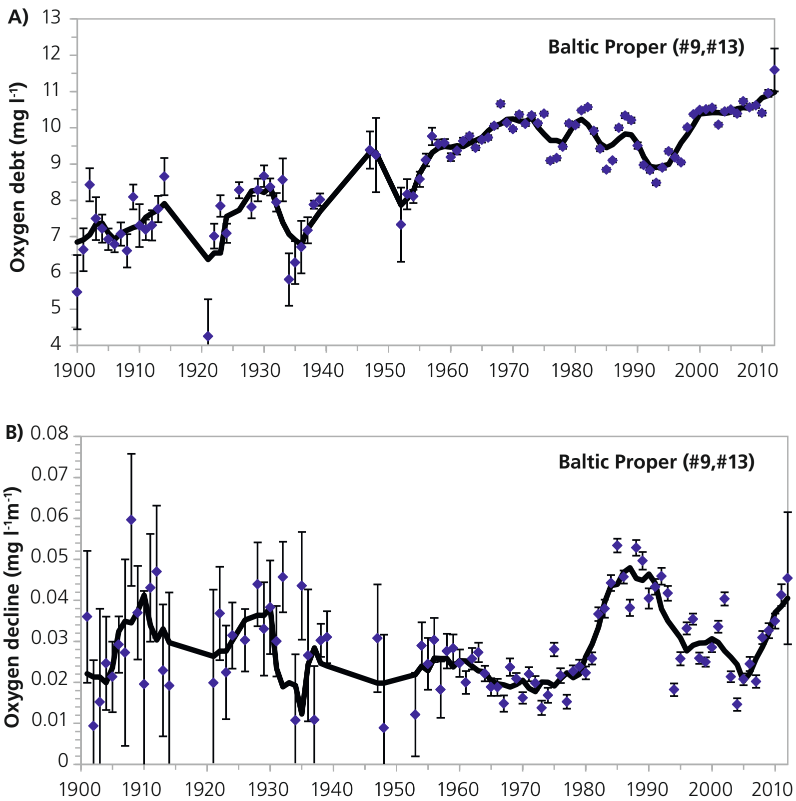


Figure 2.30 *Trends in A) oxygen debt (mg l^{-1}) below the halocline; and B) loss rate of oxygen with depth (mg $l^{-1}m^{-1}$). The solid line is the five-year moving average.*

Spatial integration over basins

The salt content of the Baltic Proper showed oscillations over the last 110 years (Fig. 2.31A). Overall, the total salt content was around 90-95×10^{15} g at the start and end of the time series, but between 1940 and 1985 there was approximately 10% more salt in the Baltic Proper. However, the salt content has been increasing during the last decade by more than 5×10^{15} g; in 2011, the salt content was around 95×10^{15} g. This observation is in contrast to various climate models that have predicted an increased freshening of the Baltic Sea (BACC 2008). During the stagnation period, approximately 10×10^{15} g of salt was lost from the bottom layer through deepening of the halocline. Trends in integrated salinity for the other basins showed minor fluctuations (Fig. C.5) around total salt contents of 3.5×10^{15} g in the Arkona Basin; 14.9×10^{15} g in the Bornholm Basin; 26.1×10^{15} g in the Bothnian Sea; and 5.0×10^{15} g in the Bothnian Bay (no data for the Gulf of Riga).

The total oxygen debt was around 20-25×10^{12} g before 1940, but from 1950 to 1970 it increased to over 30×10^{12} g (Fig. 2.31B). During the stagnation period, there was an almost complete recovery of the oxygen debt due to deepening and enhanced mixing across the halocline. However, after the large intrusion in 1993, the oxygen debt rapidly increased to a record of 38×10^{12} g in 2011 (2012 estimate is uncertain). The increasing oxygen consumption in the bottom layer has had substantial effect on the area and volume of hypoxia (Fig. 2.31C,D). Hypoxia was confined to a relatively small area and volume before 1950, but increased in 1970 to around 50,000 km^2 and 2,000 km^3, respectively. The stagnation period was associated with substantial reductions in both area and volume; however, since 1993 the hypoxic area has increased and is now exceeding 60,000 km^2 (Fig. 2.31C). The hypoxic volume, defined as the volume of water with oxygen concentration below 2 mg l^{-1}, is exceeding 2,000 km^3 (Fig. 2.31D). The trends of hypoxic area are consistent with those in Conley et al. (2009). It should also be observed that the hypoxic volume and volume below the halocline (setting an upper boundary for the hypoxic volume) have been approaching over time from less than 10% of the volume below the halocline being hypoxic to about 70% in more recent years.

The only other basin experiencing hypoxia, the Bornholm Basin, has seen somewhat similar trends (Annex B). The oxygen debt has increased by some 50%; the hypoxic area has increased from <2,000 km^2 to about 5,000 km^2; and the hypoxic volume has increased from 5-10 km^3 to >50 km^3. Thus, the total area of hypoxia in the Baltic Sea has been around 60-70,000 km^2 in the most recent years and the total volume has been over 2,000 km^3. The Arkona Basin and the Gulf of Riga may experience episodic hypoxia, particularly during summer; however, due to the episodic character of stratification, the long-term trend in hypoxia and oxygen debt could not be estimated. The Bothnian Sea and the Bothnian Bay do not have problems with persistent hypoxia; and although oxygen conditions are generally good, both these basins have also shown an increase in oxygen debt over the last 3-4 decades (Annex B), indicating that the organic loading of the bottom waters exceed the natural oxygen supply capacity of the basins, i.e. there is an imbalance between oxygen supply and consumption.

Separating anthropogenic influence from physical variations

The volume-specific oxygen debt, found as the oxygen debt below the halocline divided by the volume below the halocline, combined the variations in the oxygen profile parameters (Fig. 2.30) into a single time series (Fig. 2.32A,B). The volume-specific oxygen debt was relatively constant until ca. 1950 at levels about 5-6 mg l^{-1} and 7-8 mg l^{-1} for the Bornholm Basin and Baltic Proper, respectively, when it increased by approximately 2-3 mg l^{-1} in both basins (Fig. 2.32A,B). These overall trends are largely consistent with the increased nutrient input from land and atmosphere (Fig. 2.1), although declines in nutrient input from land over the last two decades are not yet apparent.

However, some of the variations in the observed volume-specific oxygen debt are caused by changes in the horizontal advection of bottom waters and vertical mixing across the pycnocline. The calculated Brunt-Väisälä frequency in the Bornholm Basin was approximately twice that in the Baltic Proper (Fig. 2.32C,D), indicating a much larger vertical mixing in the Baltic Proper. There were also oscillations in the Brunt-Väisälä

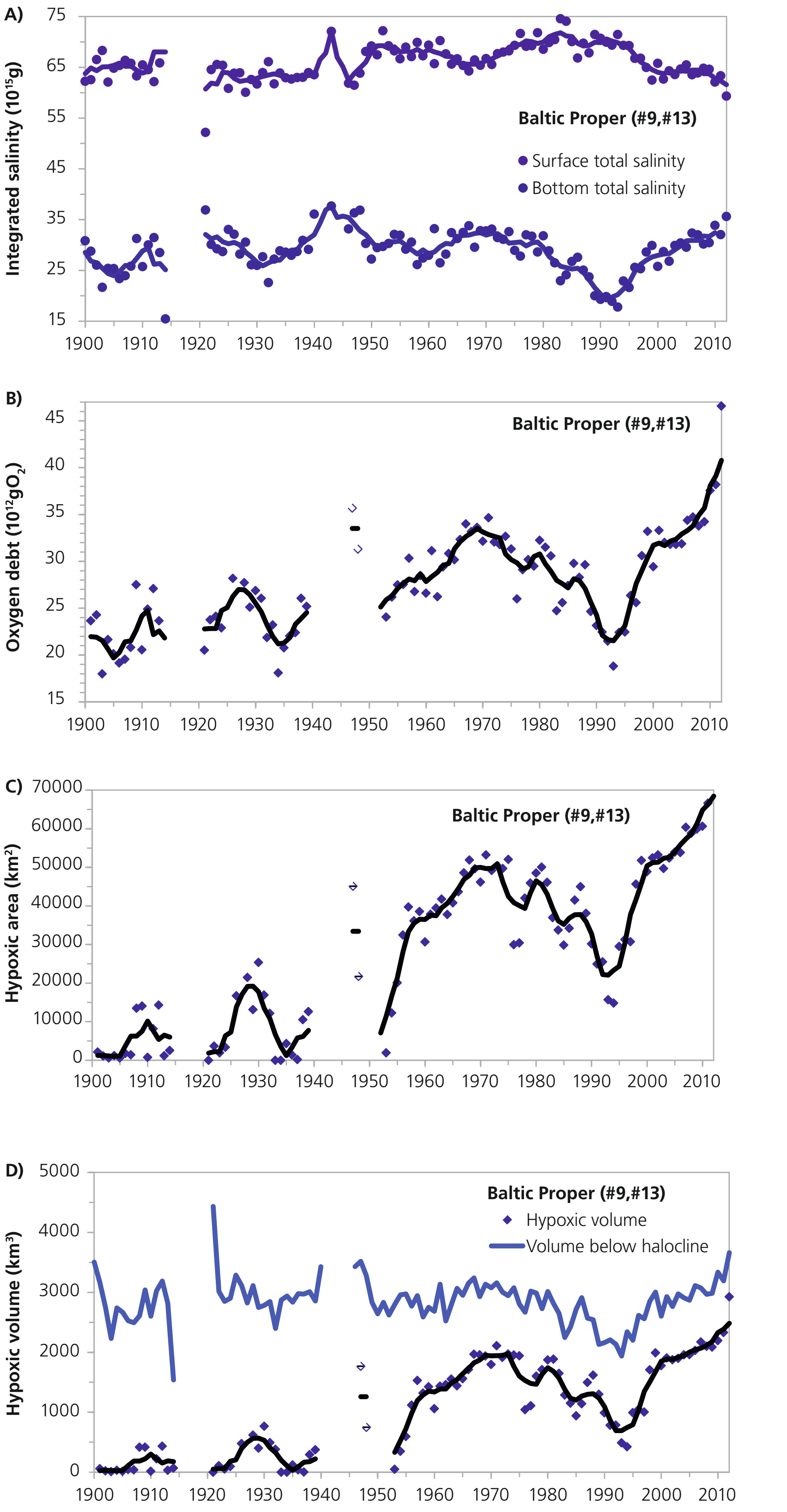


Fig. 2.31 *Spatially integrated A) salt content (10^{15} g) for surface and bottom water; B) total oxygen debt (10^{12} g O_2) for the halocline and below; C) hypoxic area (km^2); and D) hypoxic volume (km^3). The solid line is the five-year moving average.*

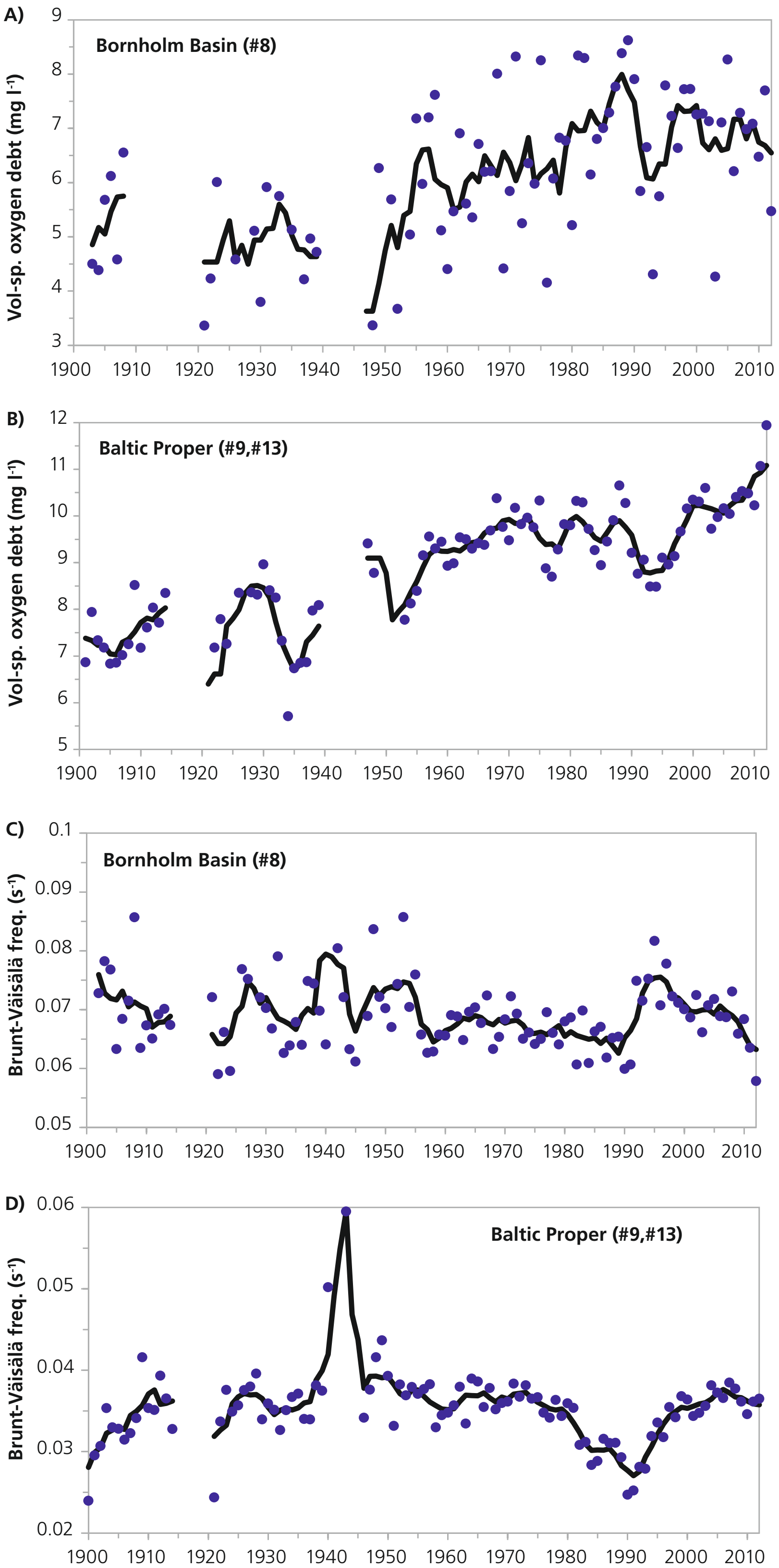


Figure 2.32 *Trends in volume-specific oxygen debt (A and B) and Brunt-Väisälä frequency (C and D) for the Bornholm Basin (A and C) and the Baltic Proper (B and D). Note the differences in scales.*

frequency matching some of the variations in the volume-specific oxygen debt, most pronounced was the decreasing Brunt-Väisälä frequency during the stagnation period from 1983 to 1993, indicating a tendency for increased vertical mixing which was reflected as decreased volume-specific oxygen debt during that period. A spike in the Brunt-Väisälä frequency was also observed in the early 1940s in the Baltic Proper; however, as the number of profiles from this period is very limited, the estimate is thus highly uncertain.

As the nitrogen and phosphorus inputs were highly correlated over time (Fig. 2.1), it was not possible to distinguish between the two nutrients in Eq. (2.2) (see Section 2.4.1). Consequently, nitrogen and phosphorus models were estimated separately for the Bornholm Basin and the Baltic Proper (Table 2.2). One-year lagged input variables were not significant in the models and therefore not shown. Similarly, Major Baltic Inflows (MBIs) did not significantly explain the variations in the volume-specific oxygen debt for any of the four models. This is likely due to the changes in bottom water salinity provide a better overall descriptor for the horizontal supply of oxygen, rather than just considering the major inflows alone. The cumulative effect in the volume-specific oxygen debt (given by the AR(1) parameter in Table 2.2) showed that there was a higher memory effect in the Baltic Proper (~10-25 years) than in the Bornholm Basin (~2-3 years), assessed as the number of years for a perturbation to be reduced to less than 10%. The models (Eq. 2.2) explained 60-65% of the variation in the Bornholm Basin and 78% of the variation in the Baltic Proper, with residual standard errors of ~0.90-0.95 mg l^{-1} and ~0.60 mg l^{-1} in the Bornholm Basin and Baltic Proper, respectively. In the Bornholm Basin, the model using nitrogen for nutrient input was somewhat better than for phosphorus, whereas there was virtually no difference between using nitrogen or phosphorus for the Baltic Proper.

It was possible to subtract the variations explained by the horizontal transport and vertical mixing of oxygen using the estimated models (Fig. 2.33). Thus, the remaining time series included the effect of nutrient input, the cumulative effect by the autoregressive factor and the residual variation. In the Bornholm Basin, the volume-specific oxygen debt, with physical forcing filtered out, increased

Table 2.2 *Model results from estimating Eq. (2.2) for the volume-specific oxygen debt in the Bornholm Basin and Baltic Proper using either total nitrogen or total phosphorus as nutrient input. Parameter estimates (see Eq. 2.2 in Section 2.4.1) are given with their standard errors in parentheses with the P-value for testing the significance below the estimates. The coefficient of determination and the residual mean standard error for each of the four models is also given. Significant parameters are highlighted in bold.*

Parameters	Bornholm Basin		Baltic Proper	
	Nitrogen	Phosphorus	Nitrogen	Phosphorus
a_0	221.5 (±32.1) (<0.0001)	218.5 (±36.4) (<0.0001)	377.1 (±36.7) (<0.0001)	333.5 (±46.1) (<0.0001)
a_N	4.66 (±0.69) (<0.0001)	8.48 (±1.57) (<0.0001)	15.77 (±5.77) (0.0076)	92.11 (±10.19) (<0.0001)
a_{MBI}	0.070 (±0.106) (0.5138)	0.140 (±0.114) (0.2243)	-0.030 (±0.108) (0.7846)	0.009 (±0.133) (0.9460)
$a_{salinity}$	11.78 (±2.25) (<0.0001)	11.10 (±2.53) (<0.0001)	17.78 (±3.77) (<0.0001)	14.27 (±4.62) (0.0027)
AR(1)	0.306 (±0.152) (0.0485)	0.354 (±0.140) (0.0134)	0.899 (±0.062) (<0.0001)	0.586 (±0.099) (<0.0001)
R^2	0.6374	0.6118	0.7750	0.7800
Residual Std. Err.	0.9129	0.9446	0.6099	0.6031

from an initial level of around 1 mg l^{-1} to around 2.5 mg l^{-1} in the 1950s, and above 3 mg l^{-1} in the most recent decades (Fig. 2.33A). In the Baltic Proper, the similarly adjusted time series increased from close to 0 mg l^{-1} to around 4.0 mg l^{-1} in the 1980s with a smaller drop in the mid-1990s to attain a present level about 3.0 mg l^{-1} (Fig. 2.33B). Thus, the most recent level in the Baltic Proper was actually lower than the level of the 1980s. The two time series had trends that were largely consistent with the nutrient inputs (Fig. 2.1), although there were differences in that the cumulative effect was small in the Bornholm Basin, suggesting that the increase in volume-specific oxygen debt was mostly a direct effect from nutrient inputs; the direct effect of the increased nutrient inputs to the Baltic Proper, on the other hand, accounted for approximately 1 mg l^{-1} and hence, the cumulative effect accounted for an additional 2 mg l^{-1}, totalling the increase in oxygen debt by some 3 mg l^{-1}.

The relatively lower oxygen debt connected with the stagnation period was likely caused by a smaller area below the halocline and consequently a relatively smaller fraction of the organic loading associated with the nutrient input affected the bottom waters, since a larger fraction of the organic material was actually sedimenting at bottoms above the halocline and therefore not contributing to the oxygen debt. Variations in the bottom water salinity accounted for up to 2.5 mg l^{-1} in the Bornholm Basin and up to 1.0 mg l^{-1} in

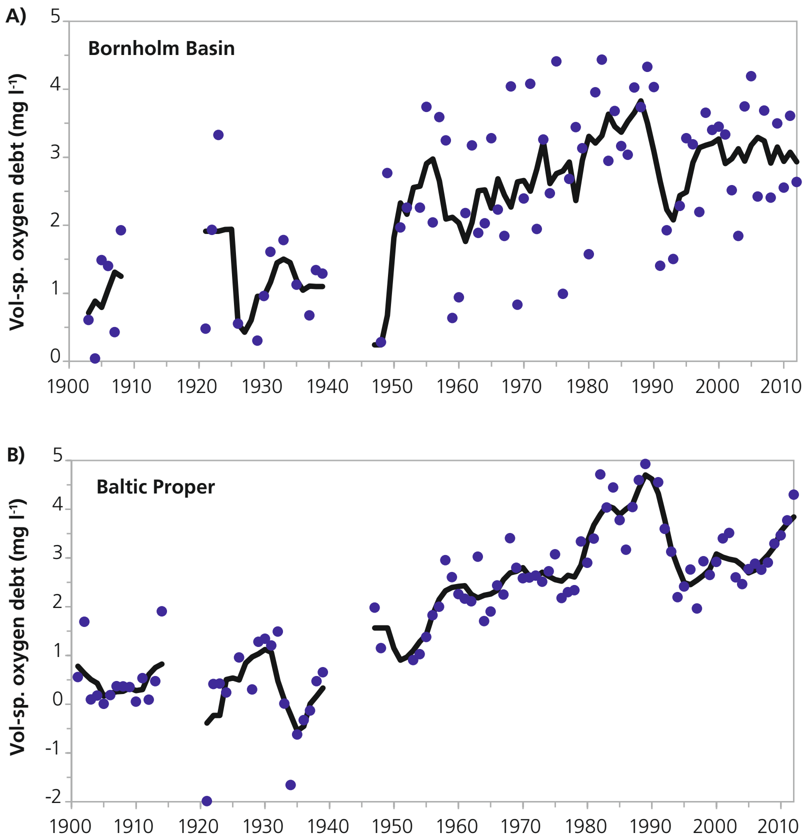


Figure 2.33 *Trends in the volume-specific oxygen debt, adjusted for variations in the horizontal transport and vertical mixing by means of the estimated models for nitrogen in Table 2.2. Similar results were obtained for the phosphorus models.*

the Baltic Proper, but there were no systematic trends in the bottom water salinity over the entire period. The modulation of the volume-specific oxygen debt by vertical mixing (as indicated by the Brunt-Väisälä frequency in Fig. 2.32) was on the order of 10-20%, but without any systematic trends over time. Thus, the only mechanism that could explain the increasing oxygen debt was the increasing input of nutrients from land and the atmosphere.

It could be argued that it is not only the direct input of nutrients that affect oxygen consumption in the Bornholm Basin and Baltic Proper, and that the models (Table 2.2) do not account for nutrient exchanges between basins or nutrient retentions in the coastal zone. This is true, and thus the models do not constitute a direct and absolute quantitative link between nutrient inputs and oxygen debt; however, the trends in nutrient inputs to the various basins have consistent patterns (Fig. 2.1) such that the interpretation of the results is valid on a relative scale. Thus, the increase by factor 3-4 of the nutrient inputs has had a direct impact on the volume-specific oxygen debt in the Bornholm Basin by approximately 2 mg l^{-1} and a combined direct and cumulative impact on the volume-specific oxygen debt in the Baltic Proper by some 3 mg l^{-1}.

2.5 Benthic fauna

In the HELCOM system of Ecological Objectives (EcoOs), benthic fauna is directly linked to the EcoO "natural distribution and occurrence of plants and animals". It has been used as one of the core indicators of eutrophication in the HELCOM integrated thematic assessment of eutrophication (HELCOM 2009) and the HELCOM Initial Holistic Assessment of the Ecosystem Health of the Baltic Sea (HELCOM 2010). In addition, it is one of the indicators of eutrophication describing good environmental status (GES Descriptor 6) in the Marine Strategy Framework Directive, as described in Commission Decision 2010/477/EU.

Soft-sediment macrofaunal communities are important components of Baltic Sea ecosystems and provide important ecosystem functions and services. These functions include the provision of food for higher trophic levels, and through the processing, reworking and irrigation of the sediments, benthic macrofauna enhance oxygen penetration and biogeochemical degradation of organic matter in the sediments. Most macrobenthic animals are relatively long-lived (several years) and thus integrate changes and fluctuations in the environment over a longer period of time. Variations in macrofaunal communities can thus be used to assess environmental conditions and disturbance events (e.g. hypoxia). Macrobenthic communities are generally food limited (Pearson & Rosenberg 1987) and the abundance and biomass of benthic invertebrates correlates to some extent with the deposition of pelagic organic material. However, as an indirect indicator of eutrophication, macrobenthos does not respond directly to increased levels of nutrients. Thus, while macrobenthic community composition provides an excellent measure of environmental status, it is more difficult to ascertain and quantify functional relationships to eutrophication. 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 distribution of macrobenthic species diversity in the Baltic Sea is limited by a latitudinal gradient of decreasing salinity and the vertical gradients of oxygen (prevalent in the Baltic Proper). Perhaps the single strongest factor influencing the biodiversity of benthic communities is the increased prevalence of oxygen-depleted deepwater. Hypoxia has resulted in habitat loss and the elimination of benthic macrofauna over vast areas.

2.5.1 Methods

The application of different indices describing the status of benthos is notoriously difficult in estuarine water bodies. For this reason, there has been a development of several different region-specific indices for coastal water bodies in the Baltic Sea. These indices, generally follow the WFD criteria, i.e. they account for benthic abundance, composition as well as the proportion of tolerant and sensitive taxa to disturbance. There are two important issues/constraints for applying these indices over broad spatial scales to the open sea areas of the Baltic Sea: (1) The dominance of a few individual species and their strong contribution to natural variations in community abundance; and (2) the problem with species sensitivity and how a particular species may be classified as sensitive in one region and not in another (see Villnäs & Norkko 2011 for an in-depth discussion).

To overcome this problem, we have adopted a more pragmatic approach for assessing the status of benthic invertebrate communities of the open sea areas in the recently completed HELCOM Eutrophication assessment (HELCOM 2009, Villnäs & Norkko 2011). The new indicator was developed to provide a harmonised assessment of benthic invertebrate status in the open sea areas across all major sub-basins. The indicator is simply based on the average benthic invertebrate diversity in a sub-basin where reference conditions and acceptable deviation have been derived utilising the best available data-set, initially for the period 1965-2006, but here updated to include the latest data available to 2008. The methods are only explained briefly here. For an in-depth description, see Villnäs & Norkko (2011).

Defining the reference conditions and assessing benthic diversity

The average regional benthic diversity - which describes the number of species in a sub-area - was used for defining reference conditions and assessing the condition of prevailing macrofaunal diversity. The measure of average regional diversity is based on point (α) diversity, i.e. the total number of species at a station per sampling occasion, including three replicates. The average regional diversity is then calculated as the average diversity of the stations in a sub-area per year (Fig. 2.34).

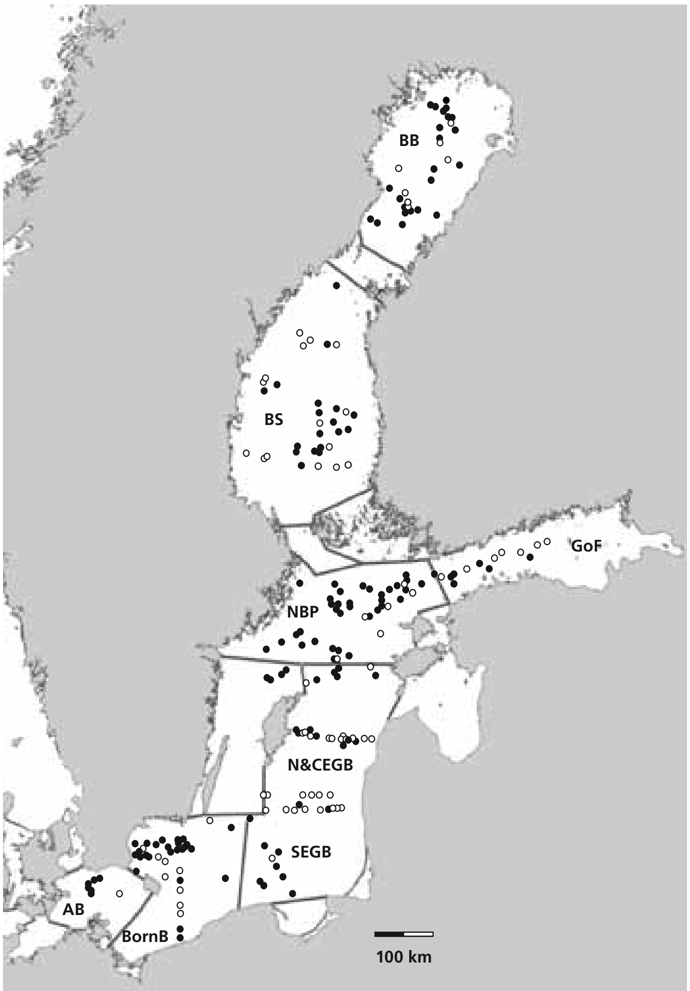


Figure 2.34 *The Baltic Sea and the sub-basins used for the analyses were: the Arkona Basin (AB); the Bornholm Basin (BornB); the south-eastern Gotland Basin (SEGB); the north and central Eastern Gotland Basin (NandCEGB); the northern Baltic Proper (NBP); the Gulf of Finland (GoF); the Bothnian Sea (BS); and the Bothnian Bay (BB). Black and white dots represent stations sampled in each area, upon which the reference values and acceptable deviation for regional species richness are based.*

Long-term monitoring data to define reference values and levels of acceptable deviation from a large number of samplings in space and time were used for the analysis (Fig. 2.34). The Baltic Sea and the sub-basins used for the analyses were: the Arkona Basin; the Bornholm Basin ; the south-eastern Gotland Basin; the north and central Eastern Gotland Basin ; the northern Baltic Proper ; the Gulf of Finland ; the Bothnian Sea ; and the Bothnian Bay. Black and white dots represent stations sampled in each area, upon which the reference values and acceptable deviation for regional species richness are based. Anoxic and/or hypoxic periods (< 2 ml O_2 l^{-1}) were excluded from the data, as were occasions when there was zero or a single occurrence of species, considered to represent initial responses to improved or impoverished oxygen conditions. The latter only occurred in areas south of the Bothnian Sea. The reference value for each sub-area was identified as the average of the 10% highest annual average regional diversity values during the monitoring period. Choosing 10% ensured that we retained a reasonable number of stations for the analysis and encapsulated at least some level of variability instead of just using the maximum value recorded. The maximum average regional diversity value was only used when the temporal data coverage was not sufficient.

Based on the long-term data used for identifying reference conditions, we defined acceptable deviation as the relative standard deviation of average regional diversity in a sub-area per year. An average acceptable deviation for each sub-area was based on data from several years. The highest acceptable deviation allowed was set to 40%, which was exceeded in the Bornholm Basin. Here we find an increased variation due to frequently occurring seasonal oxygen deficiency, which results in alternating degradation and recovery of benthic communities. In the south-eastern Gotland Basin, we also used an acceptable deviation of 40%, as there was a limited number of sampling occasions fulfilling the reference criteria within this sub-area. The Good/Moderate (G/M) boundary is defined by subtracting the acceptable deviation from the reference value.

Assessing status in benthic diversity

The prevailing status in benthic diversity can be determined as acceptable or not, if the average regional diversity values are over or under the G/M boundary identified for each sub-area (Fig. 2.34). The Baltic Sea and the sub-basins used for the analyses were: the Arkona Basin, the Bornholm Basin, the south-eastern Gotland Basin, the north and central Eastern Gotland Basin, the northern Baltic Proper, the Gulf of Finland, the Bothnian Sea and the Bothnian Bay. Black and white dots represent stations sampled in each area, upon which reference values and the acceptable deviation for regional species richness is based. Due to sparse data coverage, the assessment results for the Arkona Basin and the south-eastern Gotland Basin should be interpreted with caution.

The revised reference values and updated long-term trends in average regional benthic diversity are presented for each sub-basin.

2.5.2 Results

Benthic invertebrate diversity - and thus reference conditions - contrasts markedly between the sub-basins due to gradients in salinity, which constrains species distributions (Fig. 2.35). A total of eight basins were evaluated, in which reference conditions vary between 18.3 in the Arkona Basin and 2.1 in the Bothnian Bay. Benthic invertebrate status varies considerably between sub-basins and is related to the widespread occurrence of hypoxia and anoxia in the Baltic Proper and the Gulf of Finland. While the Gulf of Bothnia (BB and BS) has an acceptable status, the entire Baltic Proper, from the Bornholm Basin to the Northern Baltic Proper and the Gulf of Finland is in a severely disturbed state (Fig. 2.35). Data from the Arkona Basin and the south-eastern Gotland basin in particular should be interpreted with caution due to few sampling stations.

In the Bothnian Sea, a positive trend can be observed for the last few years which reflect the establishment of the invasive polychaete genus *Marenzelleria*. However, in general there are few consistent long-term trends in benthic diversity. This is mostly due to the widespread hypoxia that occurred already during the early years of the monitoring (1960s and 1970s) throughout wide expanses of the Baltic Proper. In the Gulf of Finland and the northern Baltic Proper, which has a severe problem with hypoxia, diversity maxima could be observed during the stagnation period in the mid-1990s, when oxygen conditions improved temporarily.

In the open sea areas of the Baltic Sea, benthic diversity is most strongly governed by bottom-water oxygen conditions; the quantitative links to eutrophication thus need to be examined through the presence of eutrophication-induced hypoxia/anoxia.

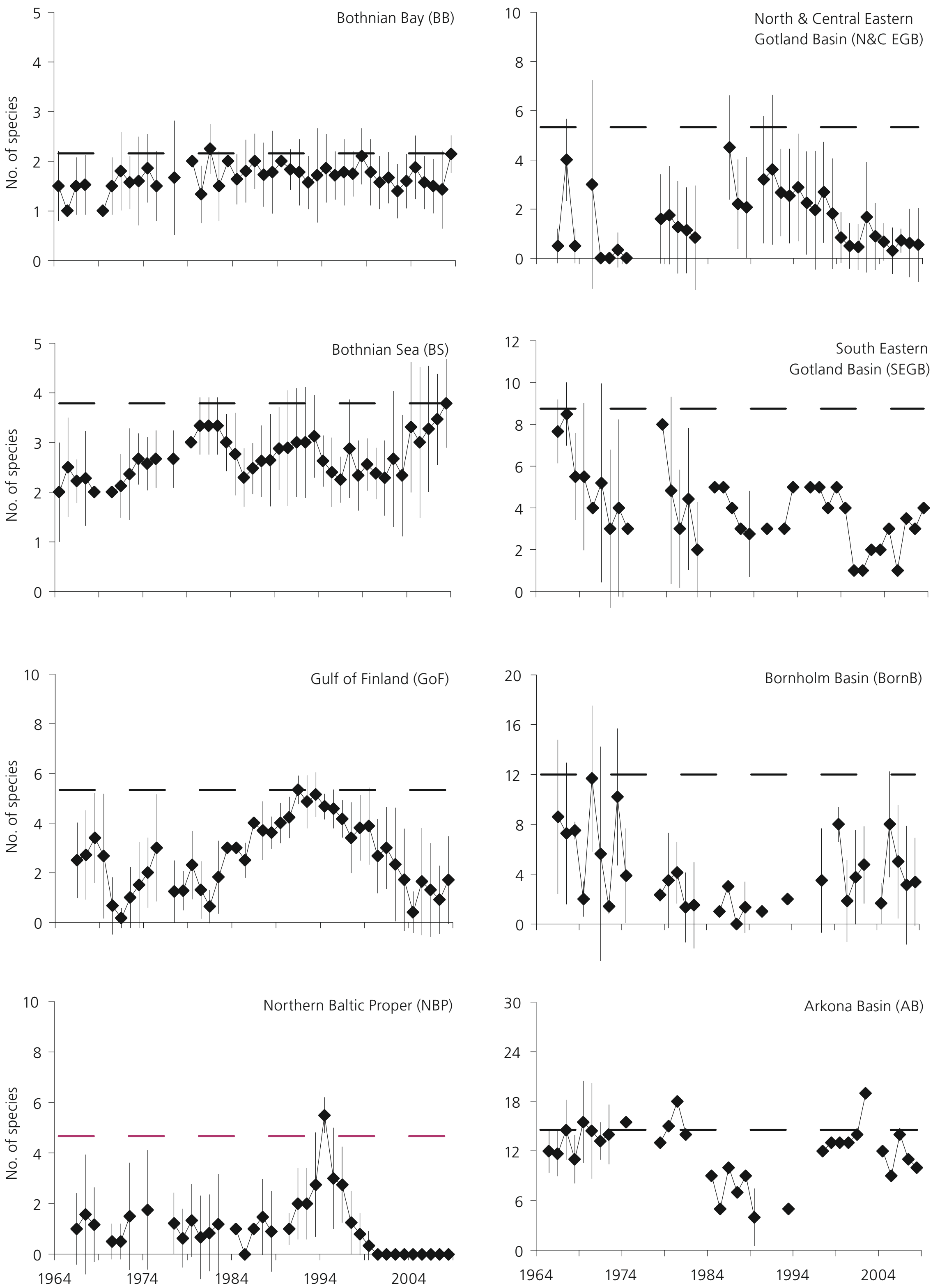


Figure 2.35 *Long-term trends in average regional diversity, based on the number of species over stations per year in each sub-basin. The dotted line indicates the reference value for the basin.*

2.5.3 Eutrophication and the enrichment of benthic biomass

Macrobenthic communities often respond predictably to organic enrichment caused by eutrophication. Pearson & Rosenberg (1978) described the qualitative relationship between softbottom macrobenthic responses to increased organic loading; several attempts have been made to quantify this widely accepted successional model, e.g. for oxygen (Gray et al. 2002) and organic carbon (Hyland et al. 2005). In Pearson & Rosenberg's model, the initial positive effects of organic enrichment on food-limited benthic communities are reflected as higher abundances and biomasses, before the community collapses at severe organic enrichment. This pattern has been well documented in coastal areas, such as the Åland archipelago (Bonsdorff et al. 1997, Villnäs et al. 2011) and in the open Baltic Sea (Cederwall & Elmgren 1990). An increased organic enrichment results in increased production, but also in increased oxygen consumption. At advanced stages of organic enrichment, the majority of bottom water oxygen is consumed by the decomposition of organic material 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.

As concluded by Cederwall & Elmgren (1990), the benthic biomass increase in areas not influenced by oxygen deficiency and the decrease in areas with low oxygen are thus two sides of the same coin. While seafloor communities in many coastal areas (typically above the halocline) exhibit signs of increased production due to enrichment effects, most of the Baltic Proper (below the halocline), experience severe oxygen depletion and a loss of biomass. Long-term data shows that these depauperate conditions were prevalent already in the 1960s when a more organised monitoring of the benthic communities commenced (Villnäs & Norkko 2011). Comparisons with data collected by Hessle in the 1920s suggest that as the major changes in benthic production may have taken place already before the 1960s, the positive enrichment effects are less clear today (cf. Cederwall & Elmgren 1990). In the Baltic Proper, including the Gulf of Finland, reduced oxygen conditions are the main reasons for low biomasses and reduced diversity.

In the open sea areas of the Gulf of Bothnia, hypoxia is not prevalent. However, also here biomass does not serve as a particularly useful measure for gauging potential eutrophication effects. Long-term data show that community abundances and biomasses vary markedly over time (Villnäs & Norkko 2011) in response to long-term population fluctuations of dominant species such as *Monoporeia affinis*. Further, species diversity on the deeper bottoms of this region is very low and patterns in community biomass over time are therefore generally driven by the population fluctuation of the few dominant species. While overall increases in production have actually occurred in many locations in the Gulf of Bothnia, this can be attributed to the introduction of the invasive polychaete *Marenzelleria* spp. and not eutrophication.

The functional relationship between increased nutrients and benthic communities is often context-dependent. Physical (e.g. water currents, turbidity, sediment structure), chemical (hydrochemical parameters, sediment composition) and biological factors as well as biotic interactions will all modify the benthic response. In the open sea areas of the Baltic Sea, low oxygen conditions are strongly limiting for benthic biomass. In areas where oxygen conditions are good, benthic biomasses generally fluctuate widely over time due to the strong population cycles of the few dominant species, making biomass as a measure of eutrophication effects less useful. This is the case for the data presented here, which covers the deeper bottoms of the open sea areas of the Baltic Sea. The story is obviously different in coastal areas, above the halocline, or in areas where slightly higher diversity is common. Here, more nuanced responses to eutrophication effects as indicated by biomass increases may readily be detected.

2.6 Ecological model simulations

Three state-of-the-art coupled physical-biogeochemical models, BALTSEM from BNI, MIKE-ECOLAB from DHI and ERGOM from IOW, were employed to simulate the response of various indicators to different load conditions: 1) a control simulation with contemporary loads (1997-2006) that represents a baseline for comparison with the two other scenarios (denoted BASELINE); 2) a pre-industrial scenario with loads representing the situation around 1900 (denoted 1900); and 3) a scenario with contemporary loads (baseline scenario) minus the reduction loads given in BSAP (HELCOM 2007a; Table 1.1) (denoted RED). Since the biogeochemical cycles are highly non-linear, we expect that nitrogen and phosphorus concentrations respond differently to the load changes and in a spatially distributed way; therefore, Chl *a*, Secchi depth and oxygen will also respond differently to load changes. For example, Savchuk et al. (2008) showed that N/P ratios were probably much larger 100 years ago in large parts of the Baltic Sea.

The models have all been validated against contemporary eutrophic conditions (Eilola et al. 2011, FEHY 2012). Eilola et al. (2011) showed that no single Baltic Sea model seems to outperform the others; rather, each model has its strengths and weaknesses. However, an ensemble of model results improves accuracy. The models used here cover a wide range in complexity, methods and resolution, which make the ensemble members rather independent.

2.6.1 Methods

All three models have performed three simulations each (BASELINE, 1900 and RED described above). For practical reasons and as a comparison with the control simulation (BASELINE), we have chosen to simulate with cyclic atmospheric forcing 1997-2006 (a ten-year period repeating itself until steady-state is achieved). The models were driven by their respective standard physical forcing, while nutrient loads were the same for all models.

An overview of the nutrient land loads of the three scenarios is given in Table 2.3. The baseline nutrient loads were compiled from the BALTSEM forcing data set (see brief description in Section 2.1 and Savchuk et al. 2012). These loads differ somewhat from the HELCOM PLC-5 compilation in that interpolation and extrapolation have been used to fill gaps in the official data (see Savchuk et al. 2012 for a discussion). The 1900 scenario loads are based on Savchuk et al. (2008). The RED scenario is a modified version of the BSAP, since it is based on the provisional reduction targets (Table 1.1) but from a modified baseline (1997-2006) as opposed to the BSAP that used the period 1997-2003. The largest differences are for phosphorus in the Gulfs of Finland and Riga, and the Baltic proper where the BALTSEM data set contains more corrections to missing data than the data set used in the BSAP.

The models were driven with a spatially more detailed distribution of the loads than given in Table 2.3; for BALTSEM, for example, the loads were aggregated onto the 13 basins (Fig. 1.2) and were also partitioned into inorganic and organic

Table 2.3 *Summary of the land-based nutrient loads (in tonnes/yr^{-1}) to the different basins in the three scenarios (1900=load situation around 1900; RED=the BSAP nutrient reductions applied to the BASELINE scenario; BASELINE=load situation 1997-2006). Total nitrogen (TN) and total phosphorus (TP) are shown in the table only, although these were partitioned into inorganic and organic nutrients, with the latter being further partitioned into bioavailable and refractory. A finer spatial distribution of nutrient loads was used for the simulations to match the spatial resolutions of the models.*

Basin	1900		RED		BASELINE	
	Phosphorus	Nitrogen	Phosphorus	Nitrogen	Phosphorus	Nitrogen
Bothnian Bay	999	36,169	2,613	54,103	2,613	54,103
Bothnian Sea	966	33,282	2,435	57,551	2,435	57,551
Gulf of Finland	3,241	63,677	6,008	97,237	8,008	103,237
Baltic Proper	4,568	166,566	6,060	218,009	18,560	312,009
Gulf of Riga	845	33,865	3,304	71,616	4,054	71,616
Danish Straits	1,070	13,800	1,269	27,191	1,269	42,191
Kattegat	825	19,567	1,570	43,577	1,570	63,577
Sum	12,514	366,926	23,258	569,282	38,508	704,282

parts. A separation between river and diffuse sources on one hand, and coastal direct sources on the other, was also done. Loads were imposed without any temporal variations, neither seasonal nor interannual cycles, and the same standard settings for bioavailability of organic nutrient loads were used for all scenarios.

Due to computational demands, the 3D models could not be run into a complete quasi-steady state for the 1900 and RED scenarios, and therefore ERGOM was run for 70 years and MIKE-ECOLAB for 50 years. However, the runs were so long that the final state could be approximated by fitting an exponential decay function to selected variables. Thus, the results from the 3D model simulations should be comparable to the BALTSEM simulations that were directly run to a quasi-steady state.

2.6.2 Model descriptions

BALTSEM

The Baltic Sea is divided into 13 sub-basins, each with its own hypsographic features (Gustafsson 2003). Horizontal contractions and/or sills separate the horizontally homogeneous sub-basins and dynamically regulate the water exchange between the sub-basins. The flow dynamics are forced by wind, varying sea levels (Carlsson 1998) and densities between the seas, and controlled by frictional resistance and dynamical flow contraction due to the Bernoulli and Coriolis effects (Stigebrandt 1990, Gustafsson 2000). The parameterisations of flows between sub-basins and through open boundaries differ due to different dynamic characteristics.

The vertical stratification is resolved by a variable number of layers, where the layers are created by inflows and kept below a maximum by fusion (Gustafsson 2000). Vertical mixing is described by a mixed layer model for the Baltic Sea (Stigebrandt 1985) and a deep-water mixing parameterisation, where the coefficient of vertical diffusion varies with the stratification (Stigebrandt 1987) and mixing wind (Axell 1998, Stigebrandt & Aure 1989). The sea-ice model follows the model of the Arctic sea ice by Björk (1992, 1997) with dynamics adapted to the Baltic Sea (Nohr et al. 2009). Heating/cooling and evaporation at the sea surface is calculated using bulk formulas (Björk 1997, Gustafsson 2003). The deep-water inflows are described by a mixing sub-model of dense gravity currents (Stigebrandt 1987). Open boundary conditions are implemented in the northern Kattegat.

The biogeochemical model (Savchuk 2002) describes dynamics of nitrogen, oxygen and phosphorus, including the inorganic nutrients nitrate, ammonia and phosphate; and particulate organic matter consisting of phytoplankton (autotrophs), dead organic matter (detritus) and zooplankton (heterotrophs). Autochthonous organic matter is produced from the inorganic nutrients by three functional groups of phytoplankton: diatoms, flagellates and others, and cyanobacteria. Organic material sinks and enters the model sediment as benthic nitrogen and phosphorus. Hydrogensulphide concentrations are represented as "negative oxygen" equivalents (1 ml H_2S l^{-1} = –2 ml O_2 l^{-1}).

Experimental setup for BALTSEM

The standard forcing used to run BALTSEM comprises atmospheric forcing variables (temperature, wind, humidity, precipitation, cloudiness and air pressure) from dynamically downscaled ERA40 data using the RCA model at SMHI (Höglund et al. 2009), daily averaged sea level observations from Hornbaek, and observations of S, T, O_2 and nutrients at the boundary to Skagerrak. River runoff was provided by SMHI (cf. Graham 1999).

For these experiments, the model was first run from standard initial conditions in 01.01.1970 to 31.12.1996; the resulting state at the end of that simulation was used as an initial condition for the experiments. For the experiments, a 300-year long forcing data set was compiled by repeating (30 times) the actual forcing 1997-2006. The sea level time series was modified so that the transition from the end of the repeated period (2006) to the beginning of a new repeated period (1997) became smooth to avoid the risk of extreme inflows. Nutrient loads from the atmosphere and land were kept constant in time.

ERGOM

The physical part of ERGOM is based on the GDFL Modular Ocean Model (MOM3) (Pacanowski et. al. 2000). Equations of motion are discretized to a z-level grid with 77 vertical layers and a horizontal resolution of approximately three nautical miles. A third-order advection scheme (Leonard 1979) is used for momentum advection, except for some critical areas near river mouths where a first-order positive definite scheme is applied. Vertical diffusion is parameterised by the KPP mixing scheme (k-profile parameterization) (Large et al. 1994) and horizontal diffusion by a Smagorinsky scheme (Smagorinsky 1963). A sea ice model is applied at the sea surface, an open boundary condition in the Skagerrak.

The biogeochemical model contains ten, three-dimensional state variables (nitrate+nitrite, ammonium, phosphate, large-cell phytoplankton, small-cell phytoplankton, cyanobacteria, zooplankton, detritus, oxygen and iron phosphate) and two, two-dimensional tracers (organic sediment and iron phosphate in the sediment). It is a model specific for the Baltic Sea and includes processes such as phytoplankton uptake and mortality; dinitrogen fixation; zooplankton growth and mortality; remineralisation of detritus; settling and respiration of detritus; retention of phosphate in the sediment and its release under anoxic conditions; and denitrification in sediments and in anoxic water. It has previously been used in nutrient load reduction experiments and climate prediction simulations (Neumann et. al. 2002, Neumann 2010).

Experimental setup for ERGOM

The model was run with atmospheric forcing from the SN-REMO model (von Storch et al. 2000). Initial conditions were taken from a previous model run in January 1997. From this point, the same ten years of atmospheric forcing (1997-2006) were applied repeatedly, seven times for each of the load scenarios (BASELINE, 1900, RED). At the open boundary in the Skagerrak, the sea level was relaxed to the Smögen gauge values. Baroclinic boundary conditions (temperature, salinity and nutrient concentrations) were taken from a climatology by Janssen et al. (1999) and left unchanged for each of the scenarios. For the riverine discharges, the total loads per BALTSEM basin provided for all the three models in common were distributed to 20 model rivers. Their freshwater discharge was left unchanged, according to the 1997-2006 period.

After the 70-year runs had been completed, an analysis was made to estimate how far the values of the last run were from a steady state. For this purpose, a time series of seven decadal means of each of the considered state variables (winter DIN, winter DIP, summer Secchi depth, annual surface Chlorophyll, and dissolved bottom oxygen) was formed and an exponentially decaying function of the type $a + b\ exp(- t / \tau)$ was fitted through the seven data points. In most cases, this function fitted the model results very well, i.e. the values of the indicators were either always increasing or decreasing with a clear tendency to converge to a steady-state. In some cases, the model results did not show an increasing or decreasing trend but oscillated within a small range. In this case, we assumed that the model had already reached steady state within the first decade. Some indicators, however, showed a constant, or even progressive, increasing or decreasing tendency throughout the seven decades of the simulation.

It transpired that the results of the steady-state estimation differed for the three load scenarios. In the BASELINE run, all investigated indicators practically reached steady-state. In the RED run, most indicators went to steady state during the simulated period. In the 1900 run, however, especially DIP concentrations were continuously declining and oxygen concentrations were continuously improving throughout the simulation period; the presented DIP values may, therefore, be seen as upper estimates and the presented oxygen values as lower estimates for the real 1900 situation.

MIKE-ECOLAB

The MIKE-ECOLAB model applied for the HELCOM TARGREV study is based on the model developed for the Fehmarnbelt Fixed Link environmental studies (FEHY 2012). The model is a three-dimensional (3-D), mechanistic, combined hydrodynamic and biogeochemical model based on the MIKE 3 FM and ECO Lab modelling systems (DHI 2011a,b). The model covers the entire Baltic Sea, Belt Sea, Kattegat and part of Skagerrak. The open boundary is located in the Skagerrak between Hanstholm in Denmark and Mandal in Norway. The model mesh is unstructured with a horizontal resolution of 3-20 km in the Baltic Sea and 1-3 km in the Belt Sea; and a vertical resolution of 1-2 m in the upper 77 m of the water column and 3-50 m in the lower layers. The bathymetric data applied for the model is mainly based on topographic charts of the sea floor provided by Bundesamt für Seeschifffahrt und Hydrographie (BSH) and digital bathymetries of Danish waters provided by the Danish Maritime Safety Administration (FRV). The model has 61 model sources representing the catchment areas of the model domain.

Since the HELCOM TARGREV model scenarios are load scenarios, the hydrodynamic part of the model will be the same for all scenarios. To save CPU-time, the scenarios have been simulated in the so-called decoupled mode; i.e., the required hydrodynamic information (water levels, current components, etc.) is read from a file which has previously been generated by running the hydrodynamic model alone.

The biogeochemical model describes algal dynamics, nutrient cycling, oxygen conditions and associated processes in the pelagic and sediment phases of the system. In the pelagic phase, the model has 28 state variables describing three algal groups (flagellates, diatoms and cyanobacteria), zooplankton, detritus, inorganic nutrients, oxygen and hydrogen sulphide as well as dissolved organic nutrients and carbon (labile and refractory). The pelagic state variables are described as 3-D concentration fields varying in time due to biogeochemical and transport processes (advection-dispersion, settling, and buoyancy). In the sediment phase, the model has 11 state variables describing the pools of nutrients and carbon in the sediment. The sediment state variables are described as 2-D areal concentration fields varying in time due to biogeochemical and transport processes (nutrients and carbon are exchanged between the water column and the bottom sediment by net sedimentation and diffusion).

The biogeochemical model is initialized by applying initial concentration fields of the state variables. Further, it is needed to specify a number of model constants and forcings. The model forcings include open boundary conditions; load of organic matter and nutrients through the model sources and atmospheric deposition; photosynthetically available radiation (PAR); and wind and current magnitudes for the re-aeration process. During the Fehmarn Belt studies, the MIKE-ECOLAB model was calibrated for the period 1990-1999 and validated for the period 2000-2007 (FEHY 2012).

Experimental setup for MIKE-ECOLAB

The model was run for the three cases: BASELINE, 1900 and RED (see above). For the simulations, it was decided to apply the ten-year period 1997-2006, and for each of the three simulation cases (baseline and two scenarios) to repeat the ten-year period a number of times to approximate a new quasi-stationary situation with changing loads. A repeated ten-year simulation applies the final fields of the state variables of the previous simulation as initial fields for a new ten-year simulation; otherwise it is identical to the previous simulation. For each of the MIKE-ECOLAB simulation cases, the ten-year period was simulated five times to get as close as practically possible to a quasi-stationary response to the load scenarios. A posterior data fitting analysis of the model results showed that the fifth repetition is close (<10%) to steady state for the major part of the indicator variables. Consequently, the model results of the fifth repetition have been applied to evaluate the scenarios.

The waterborne nutrient loads in the three simulation cases have been established by basin-wise scaling of the existing model sources (which are based on the HELCOM compilations, see FEHY 2012) in order to obtain the basin-wide loads for the three simulation cases. The atmospheric loads have been applied directly according to the provided loads for each simulation case.

Apart from the loads, no changes have been made to the original model setup.

2.6.3 Results

Nutrients

The results on winter inorganic nutrient concentrations are presented in Figs. 2.36 – 2.38, and the spatial averages for each sub-basin are given in Annex C (Table C.1-2).

Results from BALTSEM

The experiments demonstrate the large increase in winter inorganic nutrient concentrations from 1900 to the present. Typically, surface DIN concentrations have doubled and DIP concentrations almost tripled which, in turn, caused a shift to lower N/P ratios, especially in the Baltic Proper and in the entrance area. Comparisons with the third simulation run (RED) show that nutrient reductions, as proposed in the BSAP, will have a substantial effect on the surface nutrient concentrations, not only in the basins receiving lower nutrient loads, but also in the Gulf of Bothnia where DIP concentrations can be expected to decrease. In the Baltic Proper, the excepted reductions are in the order of 20% and 40% for DIN and DIP, respectively.

Results from ERGOM

Winter DIN concentrations have risen significantly from the 1900 to the BASELINE simulation, especially in the Kattegat, the Danish straits, the Gulf of Riga and the Gulf of Finland. In the open basins, they show a slight increase of about 20%. Winter DIP concentrations rose about 30% in the Baltic Sea as a whole; in areas near river mouths, however, the increase was much higher. The net effect on the N/P ratio varies between offshore and coastal areas. Around 1900, it was slightly higher in the open sea areas and lower in the Kattegat. In coastal areas in general, especially near some river mouths experiencing high phosphate load increases in the 20th century, the N/P ratio was significantly higher around 1900 compared to today's values.

However, even after 70 years of simulation with the 1900 loads, the model has not fully reached steady state and the DIP concentrations are still declining. This is due, in particular, to phosphorus that is stored in the sediments of the deep basins. This phosphorus pool, derived from the initial condition (note we have initialized the model with the 1997 situation), was obviously not present around 1900 and apparently the pool was not emptied after 70 years of simulation; for this reason, the model results for DIP are biased and thus too high in the 1900 run. The change from 1900 to BASELINE can thus be regarded as a lower estimate.

According to ERGOM, the implementation of the BSAP reduction (RED scenario) will lead to nutrient reductions. Nitrate concentrations will decline all over the Baltic Sea, except for the southern coast. The reductions are pronounced in the Gulf of Riga and Gulf of Finland. Phosphate concentrations will be reduced all over the Baltic Sea, especially near the mouths of the rivers with the largest reductions. The implementation of the BSAP reductions from the baseline (1997-2006) will shift the N/P ratio towards the 1900 value. The N/P ratio will rise significantly, especially near some river mouths in the south-eastern Baltic Sea.

Results from MIKE-ECOLAB

Winter concentrations of inorganic nutrients in the model area (Fig. 2.36-37) are reduced relative to the baseline as a result of the two scenarios (1900 and RED). The 1900 scenario accounts for the largest reductions relative to the baseline, with DIN reductions of 10-50% in the open waters and 30-70% in the bays, gulfs and coastal areas. DIP reductions have a somewhat different distribution with 40-75% in the central and eastern Baltic proper (largest at the eastern coasts and embay-

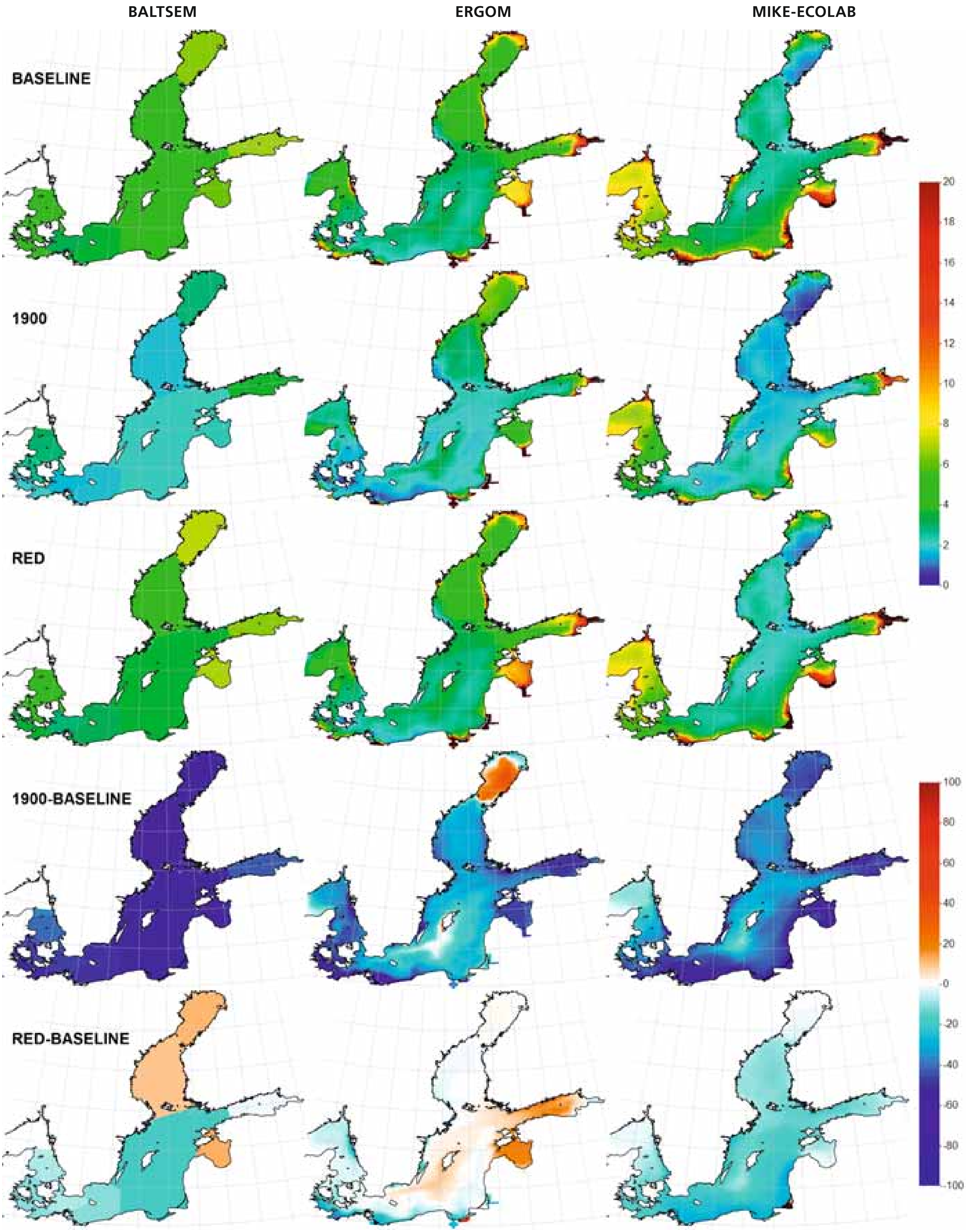


Figure 2.36 *Winter (Dec-Feb) surface DIN concentration (upper 10 m) in the BALTSEM, ERGOM and MIKE-ECOLAB models (left to right). Concentrations are given in µmol l^{-1}; changes are relative to the baseline values in (%). BASELINE=loading 1997-2006, 1900=loading around 1900, and RED=BASELINE with load reductions from the BSAP.*

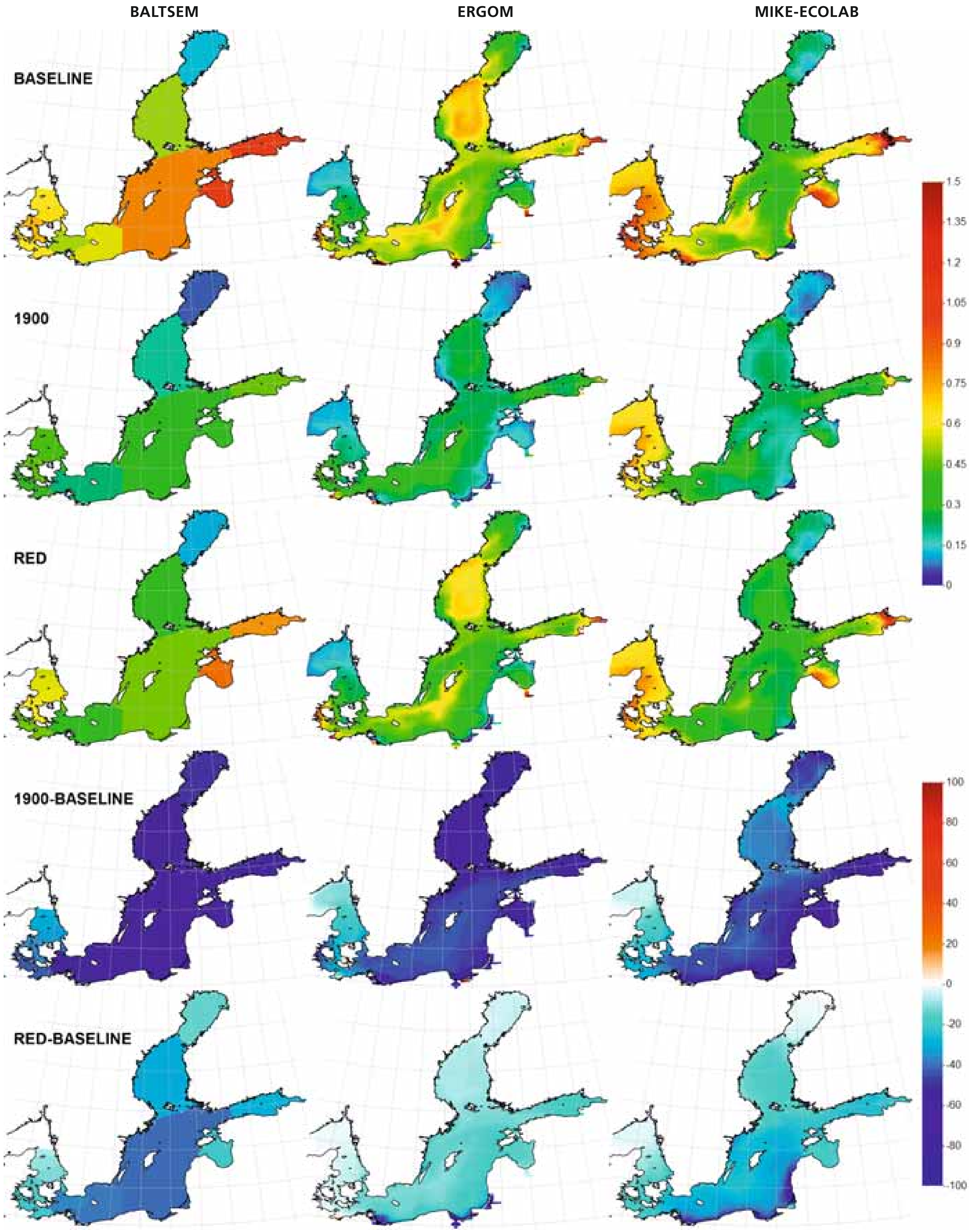


Figure 2.37 *Winter (Dec-Feb) surface DIP concentration (upper 10 m) in the BALTSEM, ERGOM and MIKE-ECOLAB models (left to right). Concentrations are given in µmol l[-1]; changes are relative to the baseline values in (%). BASELINE=loading 1997-2006, 1900=loading around 1900, and RED=BASELINE with load reductions from the BSAP.*

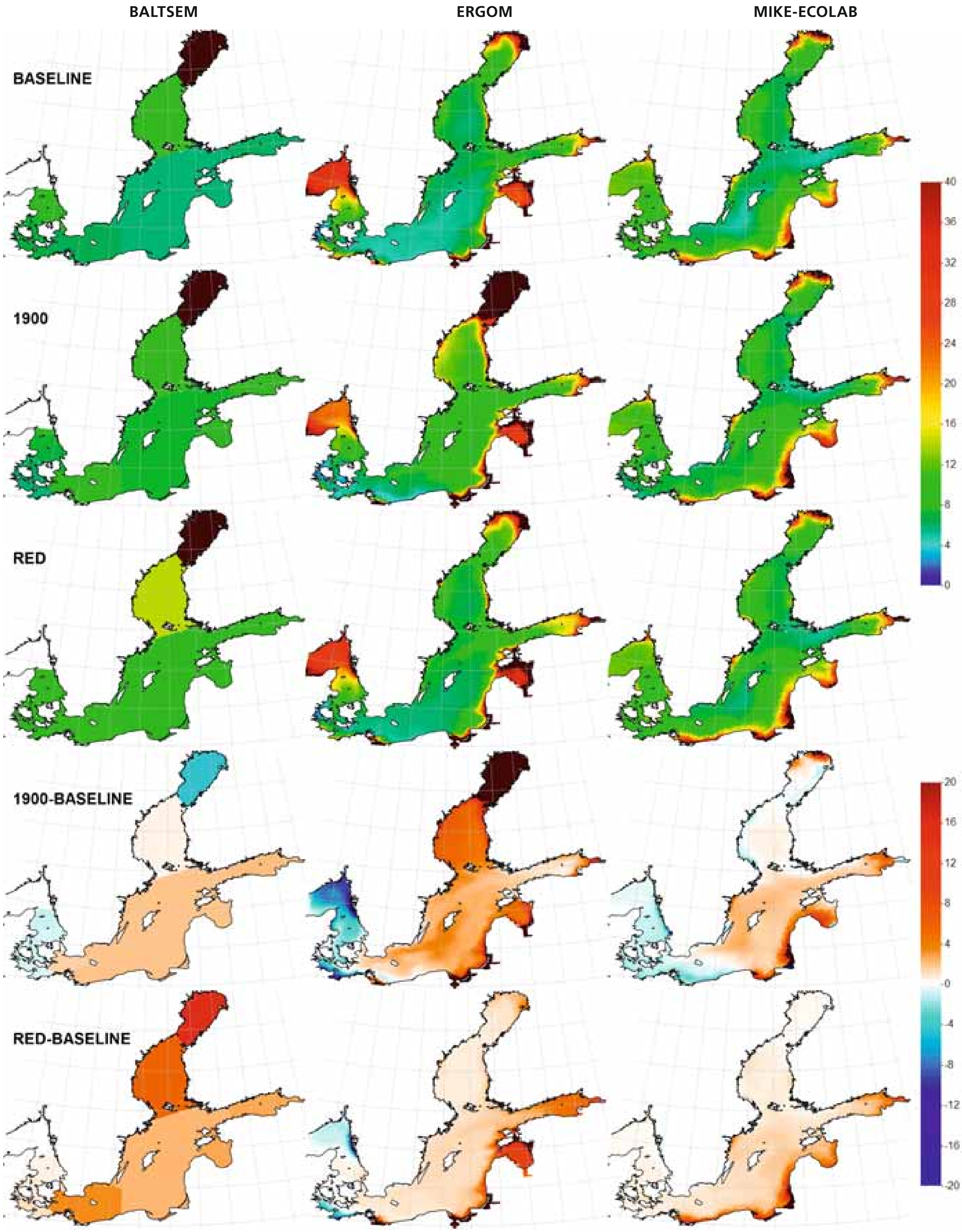


Figure 2.38 *Winter (Dec-Feb) surface DIN/DIP ratio (upper 10 m) in the BALTSEM, ERGOM and MIKE-ECOLAB models (left to right). Ratios are given as molar ratios (dimensionless); changes are in absolute values. BASELINE=loading 1997-2006, 1900=loading around 1900, and RED=BASELINE with load reductions from the BSAP.*

ments) and 10-40% in Kattegat and the Belt Sea. The RED scenario demonstrates significantly smaller reductions consistent with the smaller load reductions in this scenario. DIN reductions of 0-35% and DIP reductions of 0-60% are observed with the largest reductions in the south-eastern part of the Baltic Sea.

The uneven reductions in DIN and DIP consequently change the DIN/DIP ratio in both scenarios. The 1900 scenario shows an increasing DIN/DIP ratio relative to the baseline (Fig. 2.38) in the eastern, northern and central parts of the Baltic Sea, and a decreasing DIN/DIP ratio in the western part. The BSAP scenario shows an increasing DIN/DIP ratio in the whole model area.

Algal biomasses

The average algal biomasses in the upper 10 m, quantified as Chl *a* concentrations using Redfield ratios and C/ Chl *a* ratios between 30 and 60 (for some model varying between different algal groups), are shown in Fig. 2.39. Basin-wise averages are presented in Annex C (Table C.3).

Results from BALTSEM

Annual average Chl *a* concentrations, calculated from a fixed ratio between nitrogen and Chl-a content in algae, show a drastic increase since 1900, especially in the southern Baltic and in the Gulf of Finland, where concentrations were about 80% less than present, as well as in other parts of the Baltic Sea, which clearly suggests that nutrient enrichment has indeed caused increased algal biomasses. Nutrient reductions, as stipulated in the BSAP (RED scenario), will have a significant positive effect in reducing algal biomass. However, it seems that the effect, in general, is less pronounced in the Gulf of Finland than in the Baltic Proper.

Results from ERGOM

Chl *a* concentrations are calculated in a very simple way in ERGOM using a fixed conversion factor. Therefore, the changes shown for Chl *a* are a direct measure of the changes in the phytoplankton biomass.

Phytoplankton biomass was lower by about 50% all over the Baltic Sea around 1900. As Chl *a* concentrations, just like DIP, have not completely run into steady state after 70 years of simulation, the changes can be regarded as a lower estimate. The implementation of the BSAP will lead to Chl-*a* reductions across the Baltic Sea, but only at a magnitude of around 10%. However, Chl *a* reductions in the range of 30% are more likely near rivers carrying large loads of nutrients.

Results from MIKE-ECOLAB

The modelled annual Chl *a* concentrations are reduced as a result of the two scenarios. The 1900 scenario displays the largest reductions relative to the baseline with approximate reductions of 35-40% in large parts of the Baltic proper and up to 65% in gulfs, bays and coastal areas. The RED scenario displays more limited reductions in Chl *a* with approximate reductions of 15-20% in the Baltic proper.

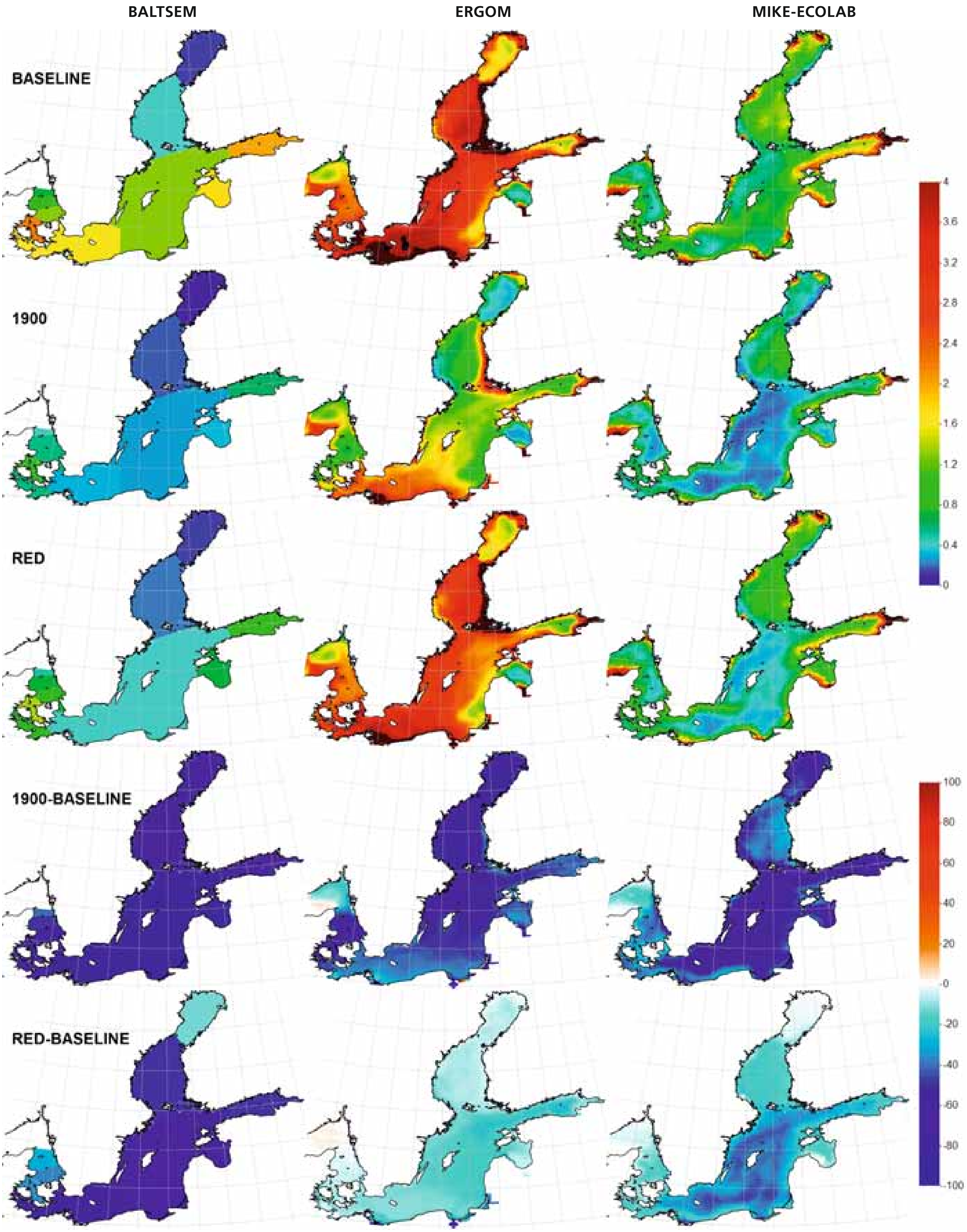


Figure 2.39 *Summer (Jun-Sep) surface Chl a concentration (upper 10 m) in the BALTSEM, ERGOM and MIKE-ECOLAB models (left to right). Concentrations are given in [µg l^{-1}]; changes are relative to the baseline values in (%). BASELINE=loading 1997-2006, 1900=loading around 1900, and RED=BASELINE with load reductions from the BSAP.*

Secchi depth

The modelled average summer Secchi depths from the three models are shown in Fig. 2.40 and the basin-wise averages are presented in Annex C (Table C.4).

Results from BALTSEM

Secchi depth in BALTSEM is computed from the biomasses of phytoplankton and zooplankton together with the concentration of detritus. In addition, a salinity-dependent relation is included to mimic the absorption by yellow substance, which is important in the Bothnian Bay in particular. The results show significant increases in Secchi depth with the reduced biomasses of the two scenarios compared to the baseline. Secchi depths in the 1900 and RED scenarios are about 10-20% and 5-13% deeper, respectively, than at present, except for the Gulf of Bothnia where only minor differences between the scenarios are found.

Results from ERGOM

In the ERGOM model, only phytoplankton and detritus are taken into account as attenuating substances. Therefore, the Secchi depth changes are correlated to the changes in Chl *a* described above. While Secchi depths were larger by about 20% around 1900, the reduction scenario (RED) will lead to minor improvements in the open Baltic Sea. Significant improvements in water transparency can only be expected in the vicinities of river mouths carrying large inputs of nutrients. However, it should be stressed that Secchi depths in the 1900 run could be underestimated since Chl *a* did not converge (see above), and generally overestimated because absorption by dissolved organic material is not included, particularly in the Gulfs of Finland and Bothnia.

Results from MIKE-ECOLAB

In the MIKE-ECOLAB, Secchi depth is modelled as a function of phytoplankton, detritus, dissolved organic substances and background extinction. The 1900 scenario shows an improvement (increase) in Secchi depth of about 20-25% in the Baltic proper relative to BASELINE, while the RED scenario shows a more limited improvement of around 10% in the Baltic proper, both consistent with the reductions in Chl *a* as mentioned above.

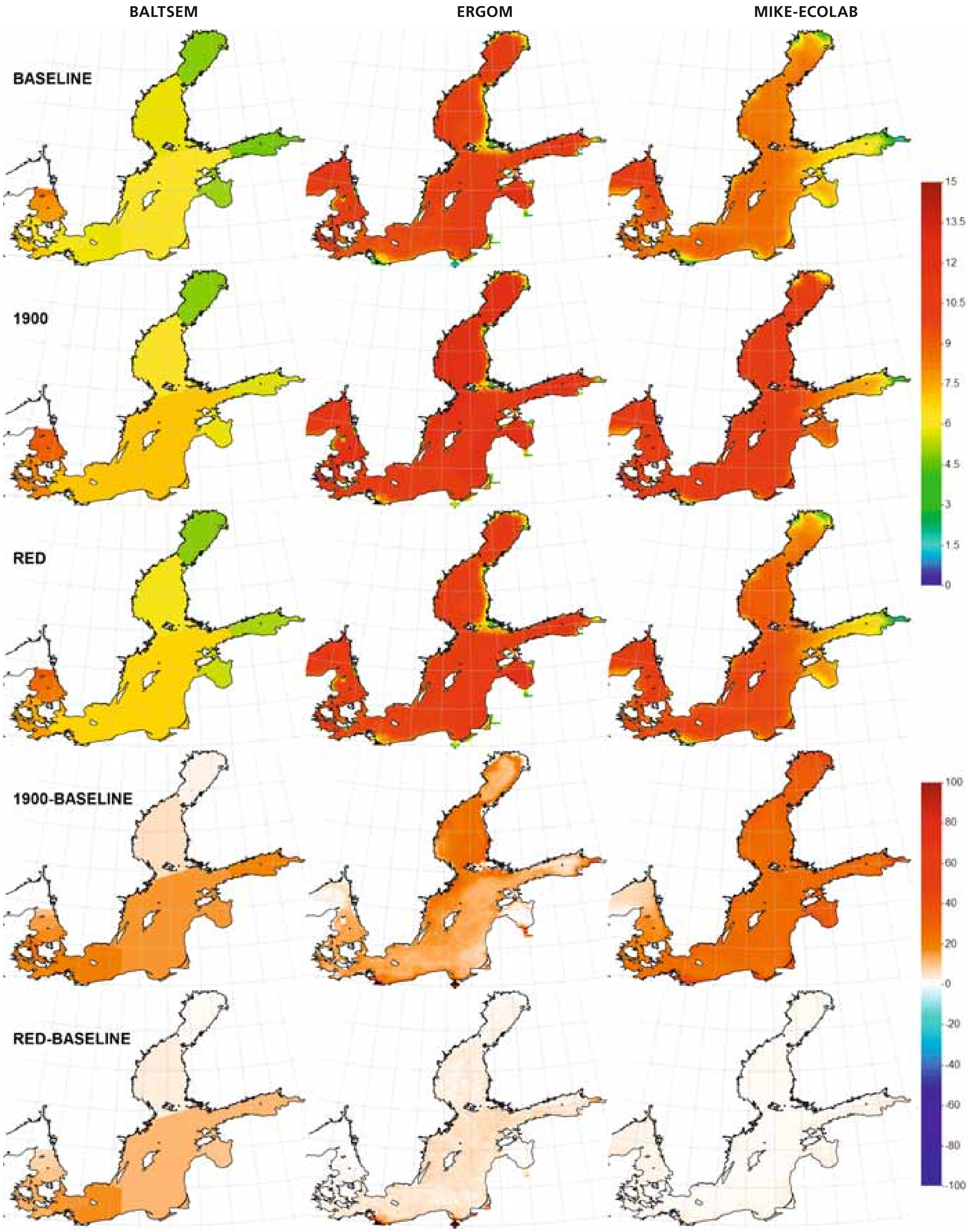


Figure 2.40 *Summer (Jun-Sep) Secchi depth in the BALTSEM, ERGOM and MIKE-ECOLAB models (left to right). Secchi depths are given in [m]; changes are relative to the baseline values in (%)*. *BASELINE=loading 1997-2006, 1900=loading around 1900, and RED=BASELINE with load reductions from the BSAP.*

Oxygen

The predicted average bottom oxygen concentrations and average oxygen profiles in the Gotland Deep from the three models are shown in Fig. 2.41 and 2.42, respectively.

Results from BALTSEM

The oxygen concentrations in the BALTSEM scenarios indicate a large decline from the 1900 scenario to the present day extensive anoxia. The RED scenario gives a significant improvement with oxygen concentrations above 2 mg l^{-1} in larger portions of the deep waters of the Baltic Sea.

Results from ERGOM

According to our model, oxygen concentrations have significantly declined from 1900 to the present. Hydrogen sulphide is taken into account as negative oxygen equivalent in our model equations. Thus, negative concentrations may exist. In our 1900 run, only the deep Eastern Gotland Basin has an average oxygen concentration below zero. In the reference run (BASELINE), the anoxic regions have spread over different Baltic Sea basins. Changes in the oxygen concentration are largest at the fringes of the deep basins, where conditions shift between oxic and anoxic. A slight improvement of oxygen conditions can be expected according to our simulations if nutrient reductions (RED scenario) are implemented.

These changes are also visible in the Gotland deep oxygen profile, displaying a shift of the redoxcline from 180 m to 140 m from 1900 to BASELINE. This change is of the same order of magnitude as the interannual variability according to the model.

Results from MIKE-ECOLAB

In the MIKE-ECOLAB model, dissolved oxygen (DO) and hydrogen sulphide are included as two different state variables; this means that the DO concentration cannot fall below zero. The modelled bottom oxygen concentrations suggest improving conditions in the two scenarios (1900 and RED) relative to the BASELINE run. The area with hypoxia (DO < 2 mg l^{-1}) is significantly reduced from the BASELINE to the 1900 scenario, although the model still predicts a considerable spread of hypoxia. The RED scenario also displays a reduction in the hypoxic area from the BASELINE, but to a smaller extent than the 1900 scenario.

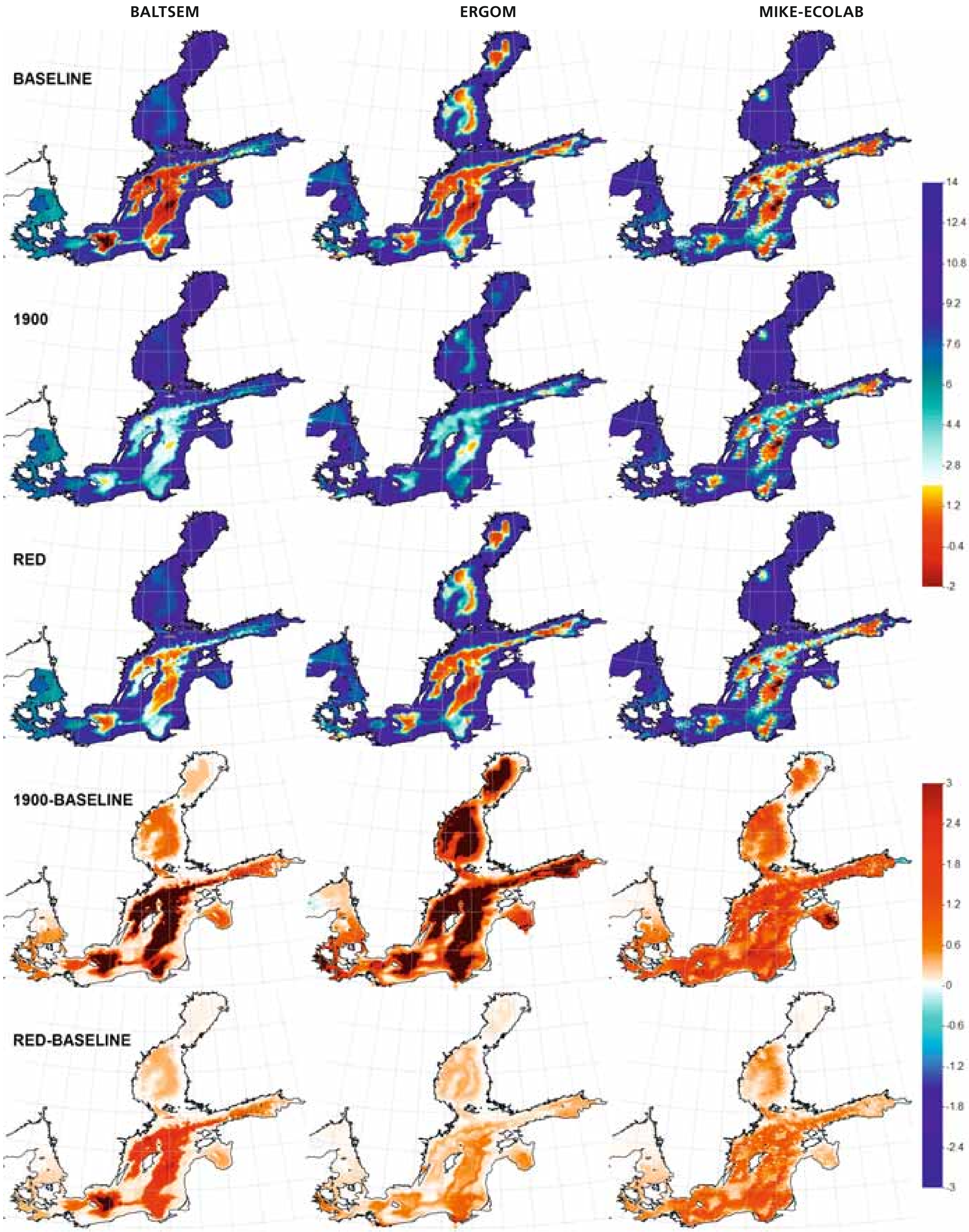


Figure 2.41 *Ten-year averaged bottom oxygen concentration in the BALTSEM, ERGOM and MIKE-ECOLAB models (left to right). Dissolved oxygen concentrations and absolute changes are given in (mg l^{-1}). In the BALTSEM and ERGOM models, negative oxygen concentrations represent H_2S. BASELINE=loading 1997-2006, 1900=loading around 1900, and RED=BASELINE with load reductions from the BSAP.*

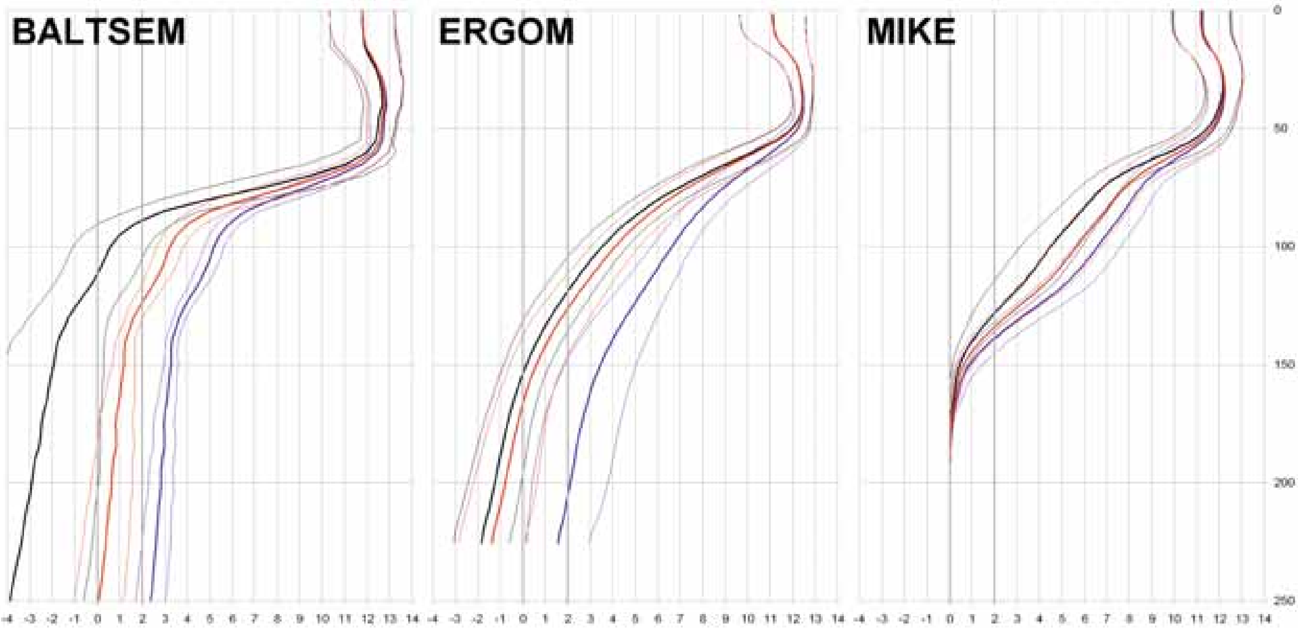


Figure 2.42 *Oxygen profiles with depth at the Gotland deep in the BALTSEM, ERGOM and MIKE-ECOLAB models (left to right). Black: BASELINE scenario. Red: RED scenario. Blue: 1900 scenario. The thick lines indicate the ten-year average; the thin lines the standard deviation. Dissolved oxygen concentrations are given in (mg l^{-1}), depth in (m).*

Summary of model results

All three models show fairly consistent responses to nutrient load changes from the BASELINE scenario. Nutrient concentrations decrease and, associated with this, Chl *a* concentrations decrease, Secchi depths increase and deepwater oxygen concentrations improve. The absolute changes in basin-averaged properties are summarised for a comparison of the model outputs (Fig. 2.43). All three models show substantially lower nutrient concentrations in the RED and 1900 scenarios compared to the BASELINE, but with some exceptions for DIN especially in the gulfs for the RED scenario. Winter DIN are on average about 2-5 mmol m^{-3} lower in the 1900 scenario, with the largest decrease in Gulfs of Finland and Riga. DIP concentration changes are quite consistent across the models, a somewhat stronger response in BALTSEM and weaker in ERGOM, while the MIKE-ECOLAB results are close to the ensemble average.

The response of Chl *a* concentration to nutrient reductions (RED) is substantially stronger in the BALTSEM results compared to the MIKE-ECOLAB and the ERGOM results. In the 1900 scenario, the BALTSEM Chl *a* concentrations are seemingly close to those obtained with the RED scenario, while MIKE-ECOLAB and in particular ERGOM predict changes in Chl *a* similar to those with BALTSEM.

One should remember that none of the three models explicitly describe Chl *a* concentrations; instead, the algal biomass is described in units of nitrogen and the conversion to Chl *a* is rather uncertain and seasonally variable.

The Secchi depth change from the baseline to the RED scenario is similar for BALTSEM and MIKE-ECOLAB, despite the fact that the change in biomass is much less in MIKE-ECOLAB, and ERGOM predicts a smaller improvement in Secchi depth. The difference between BASELINE and 1900 is similar in the BALTSEM and ERGOM outputs, except in the Gulf of Bothnia, whereas MIKE-ECOLAB suggests a much larger increase in Secchi depth across all basins. The latter was due to a change in the refractory loads that influence the supply of yellow substance in MIKE-ECOLAB, a feature not implemented in the other two models.

The models also show similar responses of oxygen concentrations to load changes.

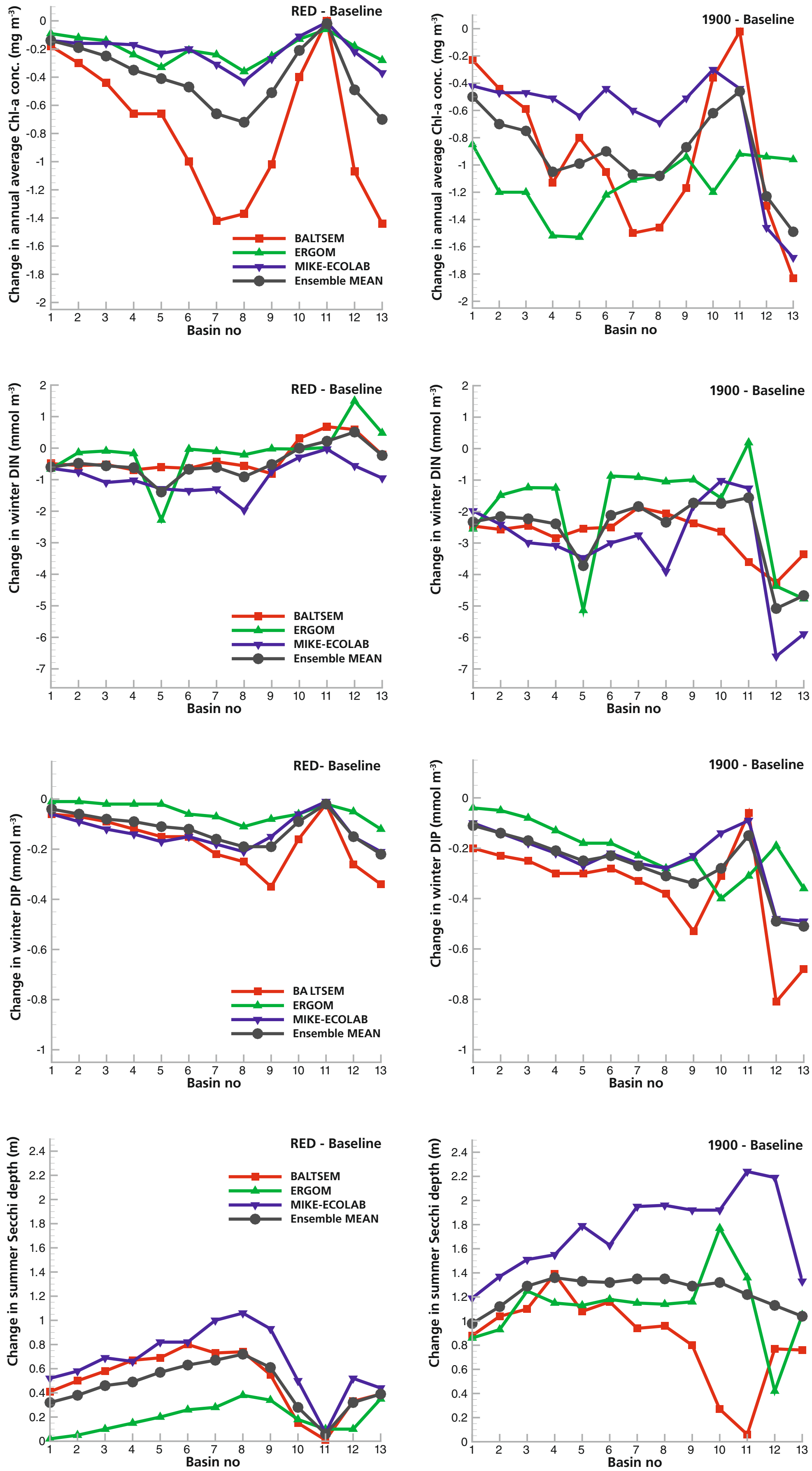


Figure 2.43 *Absolute changes of basin averages between scenarios and baseline (present day) for the different models, and the ensemble mean of the model results. The basin numbers correspond to the BALTSEM basins (Kattegat=1-3; Danish straits=4-6; Arkona Basin=7; Bornholm Basin=8; Baltic Proper=9; Bothnian Sea=10; Bothnian Bay=11; Gulf of Riga=12; Gulf of Finland=13). BASELINE=loading 1997-2006, 1900=loading around 1900, and RED=BASELINE with load reductions from the BSAP.*

2.7 Discussion and summary of trends

In this chapter, the long term trends of indicators for HELCOM's ecological objectives have been constructed from a combination of statistical analyses of the monitoring data and process-based models.

The aim was to describe the development of eutrophication-related symptoms from an anticipated yet relatively unaffected situation at the beginning of the 1900 to the present situation where most of the basins in the Baltic Sea are affected by eutrophication (HELCOM 2009). Time series for Secchi depth and oxygen are very long, in some cases more than 100 years. The series are useful for analysing both the pre-eutrophication period, the 'eutrophication period' and what in recent years might develop into an 'oligotrophication period'. Unfortunately, the time series available for nutrients, Chl *a* and benthic fauna are not as long as for Secchi depth and oxygen, and potentially may not cover the entire 'eutrophication period'.

Trends in both oxygen and Secchi depth showed a relatively stable level before the 1940s, a decade when only limited data were collected. This suggests that the Baltic Sea was relatively unaffected (compared to recent years) by human activities in this period, particularly since oxygen debt in the Baltic Proper is accumulating the effect from several decades of organic loading. Thus, if the organic loading of the bottom waters had exceeded the natural capacity of oxygen supply to the system, this would have been reflected in a gradual increase in oxygen debt, which has not been observed. After 1950, both oxygen and Secchi depth showed signs of increasing eutrophication that continued until the 1980s - and for a few basins even later. These observed trends are consistent with the modelled nutrient inputs (Fig. 2.1), reporting a 3-5 times increase of inputs from land and the atmosphere during the same period.

Time series for benthic fauna start in 1965 and nutrient and Chlorophyll monitoring began shortly after. Thus, the earliest monitoring data for these parameters do not reflect an undisturbed 'pre-eutrophication' situation. Instead, the earliest data for nutrients, Chlorophyll and benthic fauna represent a 'mid-point' in the development of eutrophication. As such, while the earliest data cannot be used to define targets for an ecosystem unaffected by eutrophication, it does possess value as an upper boundary for the validation of suggested targets in the sense that the targets should not be set for an ecological status worse than the situation in the late 1960s and early 1970s.

Reference conditions and targets have been proposed for benthic species diversity following the work of Villnäs & Norkko (2011). This approach is not comparable to the approach pursued for the other ecological objectives; however, there will be a need to confirm that the proposed targets are in line with other indicators, particularly through linking species diversity and biomass to site-specific long-term trends in oxygen and salinity. This has not been possible within the context of the present study.

The analysis of oxygen debt has provided the most comprehensive time series over the last 110 years for the Bornholm Basin and the Baltic Proper, suitable for defining oxygen status in the relatively unaffected period before 1940. Hypoxia is a major ecological problem in both of these basins. A long-term trend of oxygen debt has also been constructed for the Bothnian Sea, showing that the volume-specific oxygen debt has increased by ~1 mg l^{-1}. However, oxygen conditions are still far from ecologically critical levels in the Bothnian Sea, but they do indicate an enhanced organic loading of the bottom waters.

Fortunately, long-term trends are available for several of those basins where long-term trends of oxygen debt could not be constructed. The Danish straits, the Kattegat, and the Arkona Basin have reasonable data for between 1900 and 1920, which allows the expected water transparency during a period relatively unaffected by eutrophication to be determined. The Gulf of Finland, the Bothnian Sea and the Bothnian Bay also have 110-year long time series of Secchi depths; however, declining trends in these basins are not solely related to eutrophication as they are also associated to the enhanced delivery of dissolved organic material from land, a consequence of changes in land use and climate effects.

As the three-dimensional models were not capable of reconstructing entire time series due to the computing requirements, experiments with specific loads were performed. The most relevant for the target setting is to consider the different models' ability to predict levels of oxygen and Secchi depth around 1900. Assuming that the models were to produce results similar to what has been computed from the statistical analyses, it could then also be reasonable to assume that the models will produce reliable estimates for those parameters where historical data from that period are not available. However, such a comparison is difficult since the variation between the model results can be considerable, especially for Secchi depth due to inherent model differences and due to the lack of convergence for ERGOM in the 1900 scenario. The deviation between model results occasionally exceeds the range of variation produced by the time series. A problem for Secchi depth is that changes in the input of dissolved organic material from land is unknown and cannot therefore be reliably hindcasted by the models, which may affect hindcasts of Secchi depths, particularly for the Gulfs of Bothnia, Riga and Finland (cf. Section 2.3). The model results for 1900 fitted estimated Secchi depth levels well in the Kattegat (Annex A), whereas deviations were larger for the other basin. One problem is that the range in the model results is large, from about 1 m in the Kattegat to ~5 m in the Gulfs of Finland and Bothnia because of the level of complexity in describing the effect from DOM on light penetration. MIKE-ECOLAB has an explicit function of CDOM; BALTSEM uses salinity as a proxy for the content of CDOM; while ERGOM does not have any description on the effect of CDOM on light penetration. The large change in Secchi depth in the MIKE-ECOLAB Secchi depth results is indeed due to changes in CDOM load that was made proportionally to the nutrient load change. This probably does not represent a realistic change since for most basins, CDOM is primarily related to natural background loads; unfortunately, however, there is a lack of information on CDOM export over the last century. For the two other models, the change in Secchi depth between the present and the 1900 scenarios are similar, except for Gulf of Bothnia. For the area and volume of hypoxia, all models predict a present level of hypoxia between 50,000 and 60,000 km^2; moreover, BALTSEM and ERGOM estimate the hypoxia area around 1900 to be <5,000 km^2, which are results comparable to those observed from the historical profiles (Fig. 2.31). On the other hand, an expected area of hypoxia around 1900 of 30,000 km^2 is predicted with the MIKE model, which is inconsistent with the statistical analyses and the two other models. In summary, there are considerable variations among the model results and large variations between the model results and estimated levels for Secchi depth obtained from the statistical analyses of the observations. This is partly due to correctable inadequate parameterizations, e.g. Secchi depth variations in BALTSEM, and partly due to errors in the design of the simulation experiments. As a consequence, the present model ensemble fails to give estimates on a potential reference situation around 1900 that can be implicitly used for target setting for nutrients and Chlorophyll *a*. However, from the experiments, it became clear that a proper simulation of Baltic Sea eutrophication can only be carried out by continuously simulating the complete period from well before 1900 until the present, since the memory of the system is quite long (~ 50-100 years). For scenarios of the future, a more complex approach for physical forcing is necessary, since it transpired that the simple repetition of the physical forcing of 1997 – 2006 caused a drift in the stratification and was also too short to give appropriate interannual variability, e.g. varying stagnation periods.

The analyses presented above constitute a rather incomplete picture of how indicators for HELCOM's five ecological objectives have developed over the last century in all the basins. However, there are historical data for oxygen and Secchi depth for all basins that can be used for target setting. The earliest data for nutrients and Chlorophyll *a* constitute an upper boundary for the targets. It should also be stressed that the ecological objective for natural levels of algal blooms is only partly covered by considering the Chlorophyll *a* level, and that future work should also address the frequency, intensity and composition of phytoplankton blooms.

3. The HELCOM TARGREV target setting protocol and its use

An interim target setting protocol with three steps has been developed in order to support the revision of the BSAP, and to support the setting of evidence-based targets for eutrophication and subsequently the revision of the BSAP nutrient load reduction scheme.

The three steps are:

Step 1: Dividing the Baltic Sea into ecologically relevant basins and sub-basins with regard to eutrophication.

Step 2: Analyses of temporal trends per basin or sub-basins identified in Step 1 and the identification of any thresholds.

Step 3: From thresholds to targets – an evaluation of the ecological relevance of statistically identified thresholds.

The following sections include justifications regarding the above steps-wise procedure for eutrophication target setting for the various Baltic Sea basins, together with a thorough analysis of how the thresholds in temporal trends have been identified for different indicators and parts of the Baltic Sea.

3.1 Step 1: Dividing it all up

Dividing up the Baltic Sea into ecologically relevant units is a prerequisite, since the Baltic Sea has considerable natural gradients, e.g. regarding temperatures, ice coverage, salinity, freshwater inputs and retention times (Leppäranta & Myrberg 2008, HELCOM 2009). These features set the boundary conditions with respect to how nutrient enrichment and eutrophication is manifested in the different Baltic Sea basins. Consequently, the Baltic Sea cannot be treated as a single, uniform water mass when it comes to setting the targets – they need to be set basin-wise.

An ecologically relevant sub-division is needed in order to implement the Ecosystem Approach. The subdivision of the Baltic Sea, as outlined in Chapter 1 and used for the analyses in Chapter 2, is also comparable to what EU Member States have done when implementing the Water Framework Directive (WFD).

The subdivision used is based on the sub-division of the BALTSEM model as seen in Fig. 1.2. The BALTSEM model will be used to calculate the

maximum allowable inputs to the different basins in the revision of the BSAP. For eutrophication assessment purposes, however, the targets have been recalculated into the HELCOM sub-division using the spatial models (see Annex F).

3.2 Step 2: Time series analysis and the identification of thresholds

Once the sub-division is made, the next step is to construct basin-specific time series for the following parameters as indicators of HELCOM's ecological objectives for eutrophication: nutrients (TN, DIN, TP, DIP); phytoplankton (Chl *a*); water transparency (Secchi depth); oxygen (debt/concentration); and benthic invertebrates (diversity index). This work is the basis for subsequent steps.

In order to identify the levels of the studied physico-chemical parameters prior to excessive human influence, it is necessary to identify the time points at which the status of the eutrophication indicators changes from one level, characterised as relatively unaffected by eutrophication, to a status with significant ecological disturbance. Based on the trend analyses in Chapter 2 and the plethora of literature on changes in the Baltic Sea over the last 100 years, we propose that the period of ca. 1900 represents a status relatively unaffected by eutrophication (pre-eutrophication). This does not imply, however, that human influence was not traceable; rather, human influence was small relative to the present disturbances and may possibly represent 'natural' status, given that humans inhabit the Baltic Sea catchment and are considered part of the overall ecosystem. A significant deviation from this pre-eutrophication period, therefore, represents an altered status. Thus, we will investigate the time series derived in Chapter 2 to identify different periods of human disturbance.

Further, the knowledge of the environmental problems caused by eutrophication has improved over the last 25 years; moreover, several national and international initiatives have already been taken to reduce nutrient inputs to the Baltic Sea (e.g. Carstensen et al. 2006, HELCOM 2009). Further change points signalling a reversal of trends in later years towards earlier levels could therefore be expected.

3.2.1 Methods

The occurrence of change points in the trends of the yearly estimates of physico-chemical parameters presented in Sections 2.2-2.4 was tested by fitting piecewise linear regression models describing the level of the variable (x_t) as a function of time (year) of the type:

$$\text{For } t<k: \quad E(x_t) = \mu + \beta{*}t + \varepsilon$$
$$\text{For } t\geq k: \quad E(x_t) = \mu + \beta{*}t + \delta{*}(t-k) + \varepsilon \qquad (3.1)$$

Where *t* is time (i.e. year), *d* is the change in the slope at time *k*, and *k* is time location of the change point. This type of piecewise regression model can easily be expanded to include additional change points.

The models were fitted by optimising the likelihood function of the parameters using the PROC MODEL procedure in the software package SAS/ETS 9.2 for econometrics and time series analysis (SAS 2008). Additional information on the change point detection method can be found in Carstensen & Weydmann (2012).

3.2.2 Results

The significant change points – for example years where the slope of the linear trend changes significantly - for the physico-chemical parameters are listed in Table 3.1. For the nutrient concentrations, expressed as total nitrogen (TN), dissolved inorganic nitrogen (DIN), total phosphorus (TP), and dissolved inorganic phosphorus (DIP), the basins with changing trends in the yearly means generally showed a change during the 1980s and 1990s. The exceptions were TN in the Bothnian Sea, DIN in the Bornholm Basin and Bothnian Sea, and DIP in the Gulf of Riga, all displaying changes in the trends in the 1970s. For Chl *a*, the linear trends remained constant for all basins within the period for which data were available. Only Secchi depth in the Baltic Proper had a significantly changing slope of the linear trend in the data before 1950 (Table 3.1). The directions of the linear trends for the individual parameters in each basin are indicated by the slope of the regression line fitted to each time interval presented in Tables 3.2 – 3.7.

Table 3.1 *Significant change points in yearly levels of physico-chemical variables detected by piecewise linear regression modelling.*

Basin	TN	DIN	TP	DIP	Chl α	Secchi depth
1-3. Kattegat	1995	1995	-	-	-	-
4-6. Danish Straits	-	-	1990, 1998	-	-	-
7. Arkona Basin	-	-	-	-	-	1991
8. Bornholm Basin	-	-	1990, 1996	1977	-	-
9. Baltic Proper	-	-	1991, 1993	-	-	1931, 1984
10. Bothnian Sea	-	-	-	-	-	-
11. Bothnian Bay	1985	1975	1981	-	-	-
12. Gulf of Riga	-	-	-	1974	-	1974
13. Gulf of Finland	-	-	-	-	-	-

Change point detection for Total Nitrogen

The majority of the basins showed increasing trends in TN concentrations up to the late 1980s when concentrations stabilised at constant levels, except in the Danish Straits and the Kattegat where concentrations have been significantly decreasing since 1995-96 (Table 3.2). One basin, the Gulf of Riga, had no significant changes in the TN concentrations over the period with the data.

Change point detection for Dissolved Inorganic Nitrogen

Changes in DIN concentrations in the different basins were composed of either constant or decreasing trends (Table 3.3). DIN in the Bothnian were constant over time, whereas DIN in the Danish Straits, Arkona Basin, Gulf of Riga and Gulf of Finland continuously declined. The Kattegat and the Bornholm Basin displayed a constant level followed by a decline, whereas the Baltic Proper and the Bothnian Sea had an initial decline followed by a constant level.

Table 3.2 *Slopes of the sections of the piecewise linear regression models describing the yearly levels of total nitrogen (TN) for each basin. The arrows indicate the direction of the trend: ↗ = significantly (P≤0.05) increasing concentrations; ↘ = significantly (p≤0.05) decreasing concentrations; → = no significant changes in concentrations.*

Basin	Interval	Direction of slope	Slope	Std Err	t Value	Pr > |t|
1-3. Kattegat	1969-1995	↗	0.132	0.038	3.48	0.0012
	1996-2012	↘	-0.467	0.076	-6.15	<.0001
4-6. Danish Straits	1970-2012	↘	-0.078	0.025	-3.09	0.0036
7. Arkona Basin	1969-2012	→	0.042	0.024	1.78	0.0826
8. Bornholm Basin	1969-2012	↗	0.111	0.023	4.93	<.0001
9. Baltic Proper	1968-2012	↗	0.109	0.017	6.37	<.0001
10. Bothnian Sea	1968-2012	↗	0.035	0.017	2.08	0.0442
11. Bothnian Bay	1968-1985	↗	0.435	0.095	4.59	<.0001
	1986-2011	↘	-0.107	0.033	-3.21	0.0028
12. Gulf of Riga	1980-2011	→	-0.140	0.124	-1.14	0.2675
13. Gulf of Finland	1968-2012	→	0.023	0.034	0.67	0.5088

Table 3.3 *Slopes of the sections of the piecewise linear regression models describing the yearly dissolved inorganic nitrogen (DIN) concentrations for each basin. The arrows indicate the direction of the trend: ↗ = significantly (P≤0.05) increasing concentrations; ↘ = significantly (p≤0.05) decreasing concentrations; → = no significant changes in concentrations.*

Basin	Interval	Direction of slope	Slope	Std Err	t Value	Pr > |t|
1-3. Kattegat	1969-1995	→	-0.000	0.011	-0.04	0.9689
	1996-2012	↘	-0.065	0.026	-2.51	0.0121
4-6. Danish Straits	1971-2012	↘	-0.026	0.010	-2.88	0.0067
7. Arkona Basin	1969-2012	→	-0.004	0.005	-0.93	0.3552
8. Bornholm Basin	1969-2012	↘	-0.014	0.004	-3.25	0.0022
9. Baltic Proper	1969-2012	↘	-0.028	0.004	-6.48	<.0001
10. Bothnian Sea	1968-1975	↘	-0.267	0.052	-5.15	<.0001
	1976-2012	↘	-0.010	0.005	-2.03	0.0427
11. Bothnian Bay	1968-2011	→	0.007	0.015	0.46	0.6447
12. Gulf of Riga	1980-2011	↘	-0.280	0.046	-6.11	<.0001
13. Gulf of Finland	1969-2012	↘	-0.060	0.020	-2.96	0.0052

Change point detection for Total Phosphorus

In the majority of the basins, the TP concentrations were found to increase up until ca. 1990 when the increasing rate levelled off and concentrations even began to fall (Table 3.4); however, the concentrations began to increase again after 2000. Two basins, the Gulf of Finland and Gulf of Riga, showed increasing concentrations for the entire period, while the concentrations in the Bothnian Sea and the Kattegat did not show any significant trends, indicating constant levels of TP in these basins.

Change point detection for Dissolved Inorganic Phosphorus

Most basins showed only weak trends in DIP concentrations over the entire period, with only the Bornholm Basin and the Gulf of Riga showing changes in the linear trends (Table 3.5). In the Bornholm Basin, DIP concentrations increased until 1984; after a short period with declining concentrations, DIP levels increased again after 1997. In the Gulf of Riga, DIP concentrations remained at a constant level until 1974 and then decreased. The Bothnian Sea and the Kattegat both showed reductions in the

Table 3.4 *Slopes of the sections of the piecewise linear regression models describing the yearly levels of total phosphorus (TP) for each basin. The arrows indicate the direction of the trend: ↗ = significantly (P≤0.05) increasing concentrations; ↘ = significantly (p≤0.05) decreasing concentrations; → = no significant changes in concentrations.*

Basin	Interval	Direction of slope	Slope	Std Err	t Value	Pr > |t|
1-3. Kattegat	1932-2012	→	-0.002	0.001	-2.70	0.0098
	1969-1990	→	0.005	0.003	1.55	0.1289
4-6. Danish Straits	1991-1998	↘	-0.049	0.020	-2.49	0.0172
	1999-2012	→	0.010	0.006	1.54	0.1314
7. Arkona Basin	1967-2012	↗	0.004	0.001	3.02	0.0042
	1968-1990	↗	0.020	0.005	4.43	<.0001
8. Bornholm Basin	1991-1996	→	-0.009	0.006	-1.68	0.1015
	1997-2012	↗	0.014	0.005	2.69	0.0106
	1967-1991	↗	0.011	0.002	5.79	<.0001
9. Baltic Proper	1992-1993	→	-0.109	0.099	-1.10	0.2776
	1994-2012	↗	0.008	0.003	2.86	0.0066
10. Bothnian Sea	1967-2012	→	0.001	0.001	1.97	0.0551
11. Bothnian Bay	1967-1981	↗	-0.009	0.002	-3.56	0.0010
	1982-2011	↘	-0.003	0.001	-4.03	0.0002
12. Gulf of Riga	1973-2011	↗	0.004	0.002	2.65	0.0118
13. Gulf of Finland	1967-2012	↗	0.006	0.002	3.55	0.0009

Table 3.5 *Slopes of the sections of the piecewise linear regression models describing the yearly levels of dissolved inorganic phosphorus (DIP) for each basin. The arrows indicate the direction of the trend: ↗ = significantly (P≤0.05) increasing concentrations; ↘ = significantly (p≤0.05) decreasing concentrations; → = no significant changes in concentrations.*

Basin	Interval	Direction of slope	Slope	Std Err	t Value	Pr > |t|
1-3. Kattegat	1932-2012	↘	-0.002	0.000	-4.05	0.0002
4-6. Danish Straits	1969-2012	↘	-0.005	0.002	-2.71	0.0096
7. Arkona Basin	1968-2012	→	0.003	0.001	2.65	0.0112
8. Bornholm Basin	1968-1977	↗	0.021	0.010	2.07	0.0446
	1978-2012	→	0.001	0.001	1.12	0.2685
9. Baltic Proper	1967-2012	→	-0.001	0.001	-0.71	0.4814
10. Bothnian Sea	1967-2012	→	-0.000	0.000	-1.64	0.1079
11. Bothnian Bay	1967-2011	↘	-0.001	0.000	-5.95	<.0001
12. Gulf of Riga	1973-1974	→	0.095	0.104	0.92	0.3655
	1975-2011	↘	-0.005	0.001	-4.81	<.0001
13. Gulf of Finland	1968-2012	→	-0.000	0.001	-0.37	0.7110

concentrations for the entire period, while DIP concentrations in the Danish Straits, Arkona Basin, Baltic Proper, Bothnian Sea, Bothnian Bay, and the Gulf of Finland did not change over time.

Change point detection for Chlorophyll *a*

No change points were detected for any of the basins, indicating that the trend in Chl *a* concentration was the same for the entire period for which data were available (Table 3.6). The majority of the basins showed increasing levels of Chl *a* during the entire periods where data were available. For the Gulf of Finland and the Kattegat, however, no significant trends were detected.

Change point detection for Secchi depth

The Arkona Basin, the Baltic Proper and the Gulf of Riga all showed significant change points in the linear trends in Secchi depth (Table 3.7). Secchi depth in the Arkona Basin decreased up until 1991, after which it remained constant. In the Baltic Proper, Secchi depth increased up to 1931 and the started decreasing until 1985, after which it remained at a constant level. The Gulf of Riga showed a slightly different pattern than the two other basins as Secchi depth remained at a constant level until 1974, after which the levels have been decreasing. All other basins showed decreasing trends during the entire period for which the data were available.

Table 3.6 *Slopes of the sections of the piecewise linear regression models describing the yearly levels of Chlorophyll a (Chl a) concentrations for each basin. The arrows indicate the direction of the trend: ↗ = significantly (P≤0.05) increasing concentrations; ↘ = significantly (p≤0.05) decreasing concentrations; ➡ = no significant changes in concentrations.*

Basin	Interval	Direction of slope	Slope	Std Err	t Value	Pr > |t|
1-3. Kattegat	1974-2012	➡	-0.012	0.007	-1.86	0.0704
4-6. Danish Straits	1973-2012	➡	0.004	0.005	0.74	0.4653
7. Arkona Basin	1973-2011	➡	0.007	0.004	1.94	0.0599
8. Bornholm Basin	1973-2012	↗	0.017	0.006	2.72	0.0098
9. Baltic Proper	1972-2011	↗	0.030	0.004	7.62	<0.0001
10. Bothnian Sea	1979-2007	↗	0.029	0.013	2.29	0.0312
11. Bothnian Bay	1979-2007	➡	0.015	0.009	1.79	0.0866
12. Gulf of Riga	1990-2009	➡	-0.019	0.086	-0.22	0.8315
13. Gulf of Finland	1973-2009	➡	0.018	0.016	1.15	0.2585

Table 3.7 *Slopes of the sections of the piecewise linear regression models describing the yearly levels of Secchi depth for each basin. The arrows indicate the direction of the trend: ↗ = significantly (P≤0.05) increasing concentrations; ↘ = significantly (p≤0.05) decreasing concentrations; ➡ = no significant changes in concentrations.*

Basin	Interval	Direction of slope	Slope	Std Err	t Value	Pr > |t|
1-3. Kattegat	1906-2009	↘	-0.033	0.005	-6.00	<0.0001
4-6. Danish Straits	1903-2008	↘	-0.010	0.004	-2.39	0.0211
7. Arkona Basin	1903-1991	↘	-0.022	0.006	-3.54	0.0008
	1992-2009	➡	0.022	0.052	0.43	0.6691
8. Bornholm Basin	1903-2009	↘	-0.027	0.004	-5.96	<0.0001
9. Baltic Proper	1903-1931	↗	0.075	0.025	3.04	0.0033
	1932-1984	↘	-0.071	0.011	-6.69	<0.0001
	1985-2009	➡	-0.040	0.028	-1.43	0.1557
10. Bothnian Sea	1905-2009	↘	-0.047	0.004	-11.08	<0.0001
11. Bothnian Bay	1905-2009	↘	-0.044	0.004	-11.17	<0.0001
12. Gulf of Riga	1908-1974	➡	0.002	0.008	0.21	0.8328
	1975-2009	↘	-0.063	0.009	-7.00	<0.0001
13. Gulf of Finland	1905-2009	↘	-0.036	0.004	-9.15	<0.0001

Change point detection for oxygen debt

The volume-specific oxygen debt (Fig. 2.33), where the effects from horizontal transport and vertical mixing were filtered out, was adjusted to a mean level for the physical forcing, corresponding to a mean MBI of 9.82; mean bottom water salinities of 14.0 and 10.1; and mean Brunt-Väisälä frequencies of 0.070 s^{-1} and 0.035 s^{-1} for the Bornholm Basin and the Baltic Proper, respectively. These filtered time series, mostly representing variations linked to nutrient inputs, were analysed for change points (Fig. 3.1). In the Bornholm Basin and Baltic Proper, two change points were detected; tests for the significance of the slopes of the first and last segments of the trend curve showed that both were not different from zero ($p=0.7721$ and $p=0.8239$ for the Bornholm Basin; $p=0.5597$ and $p=0.1778$ for the Baltic Proper). Thus, the trends in both basins consisted of an initial phase with a constant level, an increasing oxygen debt over four decades, followed by stabilisation at a new elevated plateau. The change points were 1946 and 1973 for the Bornholm Basin, and 1936 and 1982 for the Baltic Proper. Overall, there was an increase in the volume-specific oxygen debt, adjusted for variations due to the horizontal transport and vertical mixing of oxygen, by 1.94 mg l^{-1} and 3.01 mg l^{-1} in the Bornholm Basin and Baltic Proper, respectively.

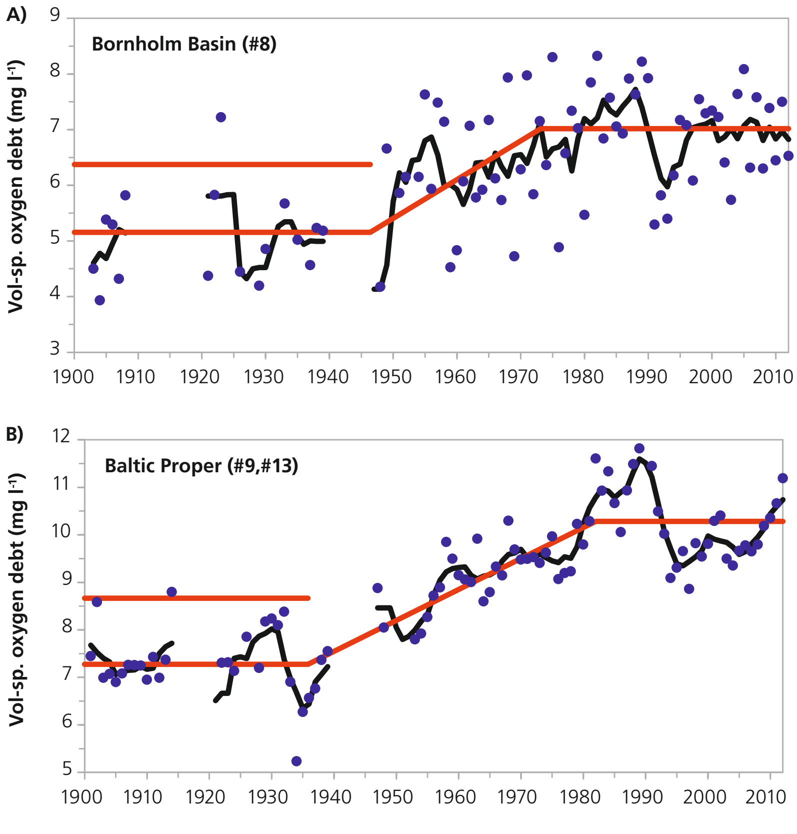


Figure 3.1 *Change point detection analysis for volume-specific oxygen debt, adjusted for variations in physical forcing by Eq. (2.2) using nitrogen as an input. The phosphorus input gave similar results. The dashed lines mark the upper 95% percentile of the oxygen debt distributions for the first period (pre-eutrophication).*

Change point detection for benthic fauna

It has not been possible to make change point analyses with regard to the abundance and biomass of benthic invertebrates.

3.3 Step 3: From thresholds to targets

The thresholds, sometimes referred to as change points in the previous section, can potentially be used for target setting. However, the ecological relevance of a potential target should be assessed and be in accordance with certain criteria. While the most ecological relevant objective is that of benthic fauna, research is still needed to link benthic fauna to oxygen conditions and other environmental factors.

A key criterion should be that the change point is identified before the onset of eutrophication on a large scale in the Baltic Sea, believed to be before the 1950s (cf. Fig. 2.1). Based on the statistical trend analyses, only oxygen and Secchi depth can be used directly for ecologically relevant target setting assuming that: 1) nutrient enrichment is the primary driver; and 2) the distribution of the eutrophication indicators in the period before the change point can be used for target setting. The model simulations have produced a 1900 scenario, which estimates a perceived 'natural' status; however, these estimates cannot implicitly be translated into targets since the simulations contain no description of the indicator distribution in such a relatively unaffected status.

Defining targets for nutrients

Targets for nutrients cannot be developed explicitly from the general approach in the target setting protocol because the time series start in the late 1960s or 1970s and are therefore not representative of what is generally regarded as a relatively unaffected or pre-eutrophication period. We can assess the nutrient levels from the earliest period with data (Annex E), acknowledging that eutrophication was already prevailing during that period, even in the open waters (cf. Fig. 3.2), and compare these with the estimates from the simulation models (1900 scenario) to infer potential targets for nutrients.

Winter DIN levels in the pre-eutrophication period were believed to be around 2 μmol l^{-1} in the central parts of the Baltic Sea and increasing towards the gulfs and the Kattegat (Table 3.8). Consistent with our perception of the eutrophication process, winter DIN levels had increased in the early 1970s. The DIN levels or preliminary targets used as background information for the BSAP in a preliminary assessment of eutrophication (HELCOM 2007b) are, however, not consistent with these results since the targets for the Kattegat are higher than the earliest data and below the simulation results for the Gulf of Finland.

Table 3.8 *Winter DIN levels (in μmol l^{-1}; Dec-Feb) estimated from the simulation models representing the pre-eutrophication period and from the monitoring data during the eutrophication period compared to the HELCOM preliminary targets that were used as background material to the BSAP. More detailed tables from the simulation models and statistical analyses are found in Annex D and E.*

Basin	Simulation models 1900 scenario			1970-1975	HELCOM preliminary targets
	BALTSEM	ERGOM	MIKE		
1. Northern Kattegat	3.07	2.49	5.82	4.75	6.8
2. Central Kattegat	2.90	2.02	5.84		
3. Southern Kattegat	2.74	1.56	4.16		
4. Samsø Belt	2.53	1.45	5.76	7.40	
5. Fehmarn Belt	1.87	2.13	3.73		
6. The Sound	2.29	1.84	3.75		
7. Arkona Basin	1.52	1.62	2.54	3.34	2.9
8. Bornholm Basin	1.67	1.77	2.90	3.29	
9. Baltic Proper	2.01	2.42	2.24	3.01	
10. Bothnian Sea	1.67	2.81	1.51	3.51	3.0
11. Bothnian Bay	2.73	6.24	1.40	6.71	5.3
12. Gulf of Riga	2.18	4.62	4.44	9.86	6.0
13. Gulf of Finland	3.54	4.63	4.93	9.22	3.8

Table 3.9 *Winter DIP levels (in μmol l^{-1}; Dec-Feb) estimated from the simulation models representing the pre-eutrophication period and from the monitoring data during the eutrophication period compared to the HELCOM preliminary targets. More detailed tables from the simulation models and statistical analyses are found in Annex D and E.*

Basin	Simulation models 1900 scenario			1970-1975	HELCOM preliminary targets
	BALTSEM	ERGOM	MIKE		
1. Northern Kattegat	0.41	0.15	0.64	0.58	0.60
2. Central Kattegat	0.41	0.19	0.66		
3. Southern Kattegat	0.42	0.20	0.56		
4. Samsø Belt	0.43	0.28	0.64	0.74	
5. Fehmarn Belt	0.33	0.40	0.60		
6. The Sound	0.33	0.25	0.45		
7. Arkona Basin	0.21	0.28	0.36	0.47	0.38
8. Bornholm Basin	0.21	0.27	0.28	0.36	
9. Baltic Proper	0.28	0.24	0.22	0.33	
10. Bothnian Sea	0.19	0.23	0.21	0.18	0.30
11. Bothnian Bay	0.06	0.09	0.10	0.06	0.15
12. Gulf of Riga	0.30	0.12	0.23	0.60	0.20
13. Gulf of Finland	0.45	0.28	0.33	0.95	0.45

Table 3.10 *Suggested targets for winter and annual means of nutrients (in µmol l^{-1}) derived as an average of the estimates of the pre-eutrophication (1900) and eutrophication (1970-75) periods for DIN and DIP, and as the estimated mean level during the early data period (1970-75) for TN and TP.*

Basin	Winter means (Dec-Feb)		Annual means			
	DIN	DIP	DIN	DIP	TN	TP
1-3 Kattegat	4.07	0.49	1.52	0.21	<17.43	<0.64
4-6 Danish Straits	5.11	0.58	1.70	0.32	<21.79	<0.97
7. Arkona Basin	2.62	0.38	1.03	0.22	<17.36	<0.66
8. Bornholm Basin	2.70	0.31	1.14	0.19	<16.29	<0.57
9. Baltic Proper	2.62	0.29	1.45	0.18	<16.23	<0.44
10. Bothnian Sea	2.75	0.19	1.57	0.13	<15.66	<0.24
11. Bothnian Bay	5.08	0.07	4.19	0.07	<16.88	<0.18
12. Gulf of Riga	<6.80	<0.41	<4.21	<0.26	<37.98	<0.71
13. Gulf of Finland	6.79	0.65	3.35	0.27	<22.15	<0.56

Similar results were obtained for winter DIP with levels for the pre-eutrophication period around 0.20 µmol l^{-1} in the central parts of the Baltic Sea and increasing towards the gulfs and the Kattegat. The winter DIP levels assessed from the early data were generally higher than pre-eutrophication estimates, except for the Gulf of Bothnia. The HELCOM preliminary targets (HELCOM 2007b) were also inconsistent with the results from the simulation models, while the statistical analyses mostly showed levels above those experienced in the early 1970s.

Recognising that nutrient levels in the early 1970s represent an ecosystem affected by eutrophication and that the 1900 scenario from the simulation models represent an average situation for the pre-eutrophication period, it is implicit that the boundary between a relatively unaffected and an effected system must be somewhere in between. However, we have no further information on where this boundary may be within this interval. A rough estimate of nutrient targets would therefore be an average of the levels estimated around 1900 and the early 1970s (Table 3.10). These targets present an advance to those in the preliminary HELCOM assessment (HELCOM 2007b), which were not consistent. It is possible that DIN and DIP levels could be potentially influenced by contamination, which is a known problem in the earlier data; however, the outlier detection employed and the consistency of the estimates in Tables 3.8 and 3.9 suggest that this potential problem does not severely affect the proposed targets. As TN and TP were not simulated in the models, it is proposed that the targets should be less than the levels reported for the early 1970s (Table 3.10). For the Gulf of Riga, as the earliest nutrient data were from the 1980s and 1990s, the approach sketched above only gives an upper limit for the nutrient levels.

Table 3.11 *Summer Chl α levels (in µg l^-1; Jun-Sep) estimated from the simulation models representing the pre-eutrophication period and from the monitoring data during the eutrophication period compared to the BSAP targets. More detailed tables from the simulation models and statistical analyses are found in Annex D and E.*

Basin	Simulation models 1900 scenario			1972-1980	HELCOM preliminary targets
	BALTSEM	ERGOM	MIKE		
1. Northern Kattegat	0.380	1.347	0.464	1.22	1.9
2. Central Kattegat	0.501	1.027	0.432		
3. Southern Kattegat	0.538	0.879	0.398		
4. Samsø Belt	0.841	1.181	0.659	1.89	
5. Fehmarn Belt	0.525	1.917	0.519		
6. The Sound	0.566	2.232	1.010		
7. Arkona Basin	0.299	2.341	0.526	1.44	
8. Bornholm Basin	0.291	2.628	0.523	2.44	
9. Baltic Proper	0.247	1.466	0.367	1.74	1.5
10. Bothnian Sea	0.140	1.066	0.590	1.52	1.5
11. Bothnian Bay	0.040	0.641	0.548	1.63	1.5
12. Gulf of Riga	0.319	0.617	0.648	4.12	1.7
13. Gulf of Finland	0.592	1.465	1.386	4.37	1.8

Defining targets for Chlorophyll *a*

Targets for Chl *a*, as for nutrients, cannot be developed explicitly from the general approach in the target setting protocol because the time series start in the 1970s and are therefore not representative of what generally is regarded as a relatively unaffected or pre-eutrophication period. The Chl *a* from the earliest data period (Annex E) is compared with estimates from the simulation models (1900 scenario) to infer potential targets for Chl *a*.

Summer Chl *a* estimates across models were quite variable, most likely due to differences across models in converting phytoplankton biomass from the different algae groups to Chl *a*. This uncertainty implies that reliable estimates of Chl *a* in the pre-eutrophication period are not available. The scientific foundation of the Chl *a* preliminary targets in the BSAP (HELCOM 2007b) is not known, but some of these targets appear reasonable compared to the level in the 1970s and others seem either too high (e.g. the Kattegat) or too low (e.g. the Gulf of Riga and Gulf of Finland). We therefore propose that Chl *a* targets are set below the estimated level of the 1970s.

Defining targets for Secchi depth

In general, only Secchi depth in the Baltic Proper showed change points that may be associated with the onset of eutrophication during the 20th century, altering the levels of the tested physico-chemical parameters. None of the other available time series analysed above extended far enough back in time to identify the potential starting point of the eutrophication process. Moreover, the time series for Secchi depth contain large gaps, making it difficult to specifically identify change points. Therefore, we will use the change points identified for oxygen debt (see below) to define the years before 1940 as a period relatively unaffected by eutrophication. The distribution of the annual Secchi depth means in this period was estimated, acknowledging that the variation in the annual means were combined of two sources of uncertainty: 1) interannual variation in mean Secchi depth; and 2) uncertainty associated with the estimation of the annual mean. The latter can be considerable since the number of observations before 1940 is rather limited, which can also be clearly seen in the trends (Fig. 2.18 and Fig. B.11). Thus, in order to quantify the magnitude of the interannual variation, a mixed model was employed where the standard error of the annual means was used to characterise the uncertainty of determination and the random variation between years was estimated.

The distribution of the annual Secchi depth means ranged from about 5 m in the Gulf of Riga to 11 m in the Baltic Proper (Table 3.12), whereas interannual variation was largest in the Bothnian Sea and Baltic Proper, and the smallest in the Arkona Basin. The means of the distributions showed the expected gradients from the Kattegat, decreasing towards the shallower Danish Straits, and then increasing with peak levels in the Baltic Proper followed by decreasing tendencies towards the three gulfs connecting to the Baltic Proper (Gulf of Bothnia, Gulf of Finland and Gulf of Riga). These spatial trends are, to a large extent, a reflection of the connectivity to land. The standard error of the distributions was largely scale-dependent, i.e. the larger the mean the larger the standard error. The only exception from this pattern was the Arkona Basin and Bornholm Basin, where the annual means were generally quite uncertain (see Annex B). As a consequence, the interannual variation could not be estimated for the Bornholm Basin (SE=0), and it is likely that the interannual variation for the Arkona Basin is underestimated. A more realistic estimate could be the average of the standards errors in the Danish Straits and the Baltic Proper, yielding a suggested interannual variation in Secchi depth means for the Arkona Basin and Bornholm Basin of 1.18.

Table 3.12 *Distribution of the annual means of Secchi depth (m) from before 1940 given as mean ± standard error. The lower 5-percentile of the distribution is found as the mean - 1.645×standard error. The BSAP preliminary targets are the annual mean employed for the BSAP that were derived from the HELCOM EUTRO project (the target for the Baltic Proper also covers the Arkona and Bornholm Basins).*

Basin	Period	Mean ± SE	5-percentile	HELCOM preliminary target
1-3 Kattegat	1906-1911	9.56±1.24	7.52	9.0
4-6 Danish Straits	1903-1912	7.88±0.94	6.33	7.7
7 Arkona Basin	1903-1912	9.10±0.48	7.16**	
8 Bornholm Basin	1903-1912	10.21±0	8.27**	
9 Baltic Proper	1903-1939	11.12±1.42	8.78	8.2
10 Bothnian Sea	1905-1939	9.28±1.45	6.89	8.1
11 Bothnian Bay	1905-1939	8.45±1.25	6.39	6.6
12 Gulf of Riga	1908-1970*	4.97±0.65	3.90	4.2
13 Gulf of Finland	1905-1938	6.78±0.83	5.41	6.0

*Including data up to 1970 because there were too few annual means before 1940. See trends in Annex A.
** Percentile calculated from a revised estimate of the standard error of 1.18

Table 3.13 *Distribution of summer means (Jun-Sep) of Secchi depth (m) from before 1940 given as mean±standard error. The lower 5-percentile of the distribution is found as mean-1.645×standard error. The EUTRO targets are summer means (Jun-Sep) derived from the work of Fleming-Lehtinen & Laamanen (2012).*

Basin	Period	Mean ± SE	5-percentile	EUTRO targets
1-3 Kattegat	1906-1909	9.47±0	7.56**	7.9
4-6 Danish Straits	1903-1909	9.00±0.76	7.75	
7 Arkona Basin	1903-1909	9.15±0.94	7.60	
8 Bornholm Basin	1903-1909	8.75±0	6.84**	
9 Baltic Proper	1903-1939	9.80±1.37	7.55	7.0
10 Bothnian Sea	1905-1939	9.22±1.48	6.79	6.8
11 Bothnian Bay	1905-1939	7.99±1.02	6.31	5.6
12 Gulf of Riga	1908-1970*	4.86±0.68	3.74	4.5
13 Gulf of Finland	1905-1936	7.02±0.97	5.42	6.0

*Including data up to 1970 because there were too few annual means before 1940. See trends in Annex A.
** Percentile calculated from a revised estimate of the standard error of 1.16.

The distribution of summer means (June-September) was calculated with the same approach using a mixed model (Table 3.13). The spatial trends showed a similar pattern to the annual means (Table 3.12) with the highest values observed in the Kattegat and the Baltic Proper, and the lowest summer means observed in the Gulf of Riga. There were even fewer yearly summer means than the annual means of Secchi depth; further, the interannual variation could not be estimated for the Kattegat and the Bornholm Basin (SE=0). Following the same procedure as above, the interannual variation for the Bornholm Basin was set to the average of the standard errors in the Arkona Basin and the Baltic Proper (SE=1.16); this value was also found suitable for the Kattegat that had a mean summer Secchi depth in the range between those of the Arkona Basin and the Baltic Proper.

The 5-percentile is the lower 95% confidence limit for the distributions and was calculated from the means and standard errors (Tables 3.12 and 3.13). Means above the 5-percentile can be considered to belong to the distribution of annual or summer Secchi depth means before 1940, which is assumed to represent a period relatively unaffected by eutrophication, whereas the annual means below the 5-percentile are unlikely to belong to these distributions characterising the natural interannual variation in Secchi depth before 1940. Thus, the 5-percentile can be regarded as the boundary between a relatively unaffected and an affected Baltic Sea, and may as such be proposed as targets for the revision of the BSAP.

These values have been compared to the targets from the EUTRO project and the BSAP (Table 3.12

and 3.13). The EUTRO targets are essentially developed on the same data set albeit from a slightly different approach, using summer Secchi depth ca. 1900 as reference conditions and setting the targets to a 25% acceptable deviation. In the BSAP, these targets were recalculated into annual Secchi depth means. Overall, the proposed targets from TARGREV are comparable to those obtained in the BSAP and EUTRO project (Fig. 3.2). The proposed targets for annual Secchi depth were generally lower than the BSAP, with the exception of the Baltic Proper where the proposed targets were larger (8.78 versus 8.2 m, cf. Table 3.12). In the Kattegat, Danish Straits and Bothnian Sea, TARGREV targets were more than 1 m below those used in the BSAP. For summer Secchi depth, the differences between the targets from TARGREV and EUTRO were small (~0.5 m), although the targets according to TARGREV suggest a strengthening by 0.55 and 0.71 m in the Baltic Proper and the Bothnian Bay, respectively; and alleviations by 0.6-0.7 m in the Gulf of Riga and the Gulf of Finland.

Although the Secchi depth targets are comparable to those previously employed, it should be stressed that the statistical approach in TARGREV is more appropriate since spatial and seasonal variations within the basins are accounted for, which was not the case for the BSAP and EUTRO targets.

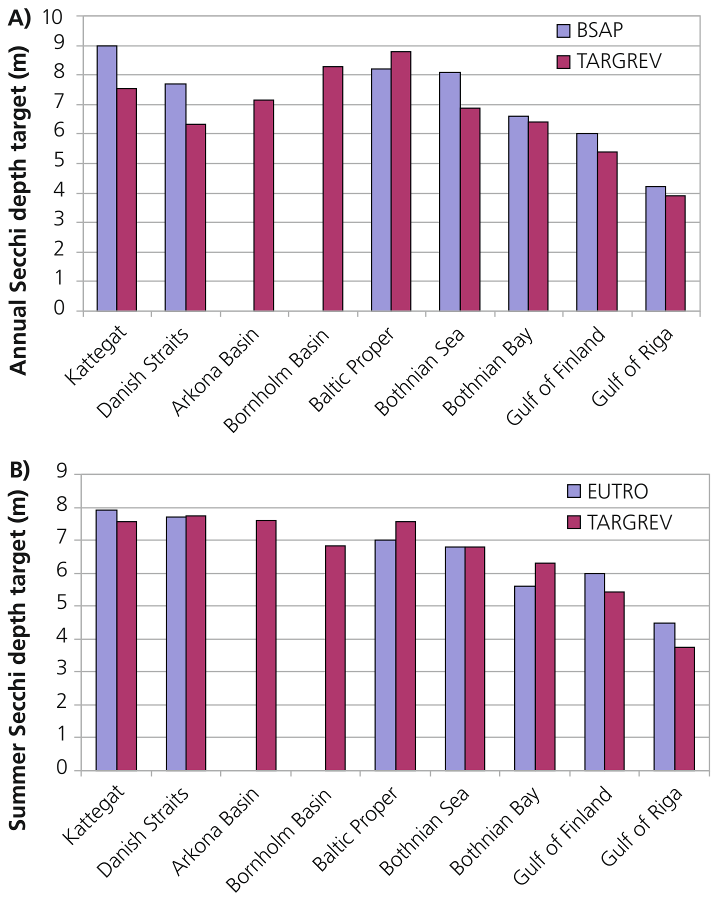


Figure 3.2 *Comparison of the proposed targets for A) annual mean Secchi depth and B) summer (Jun-Sep) mean Secchi depth with those used in the BSAP (A) and EUTRO project (B).*

Defining targets for oxygen debt

The trends in oxygen debt (Fig. 3.1) are consistent with the trends for nutrient inputs, except that decreasing nutrient inputs during the last 10-20 years have not yet resulted in a significant decline in oxygen debt, although volume-specific oxygen debt peaked in the 1980s in both basins and was lower in the 1990s and 2000s. If we consider the period before oxygen debt started to increase, then this period of around 1900 constitutes a state with minor anthropogenic disturbance, potentially qualifying as 'natural' state under recent climate conditions, acknowledging that humans inhabit the Baltic Sea catchment and should be considered part of the ecosystem. The volume-specific oxygen debt was approximately normal distributed with a mean of 5.07 mg l^{-1} and a standard deviation of 0.79 mg l^{-1} in the Bornholm Basin; and a mean of 7.25 mg l^{-1} and a standard deviation of 0.86 mg l^{-1} in the Baltic Proper. If these two distributions represent the natural variation in oxygen debt, then we can consider oxygen debts above the 95% confidence interval for the distribution as representing a significant departure from the natural variation within the two basins. Hence, following this line of argumentation, volume-specific oxygen debts, when adjusted for horizontal transport and vertical mixing, above 6.37 mg l^{-1} in the Bornholm Basin and above 8.66 mg l^{-1} in the Baltic Proper are significantly disturbed as a consequence of excess nutrient inputs. These levels should be compared to the contemporary levels for the last two segments of the trend curves, which are 7.02 mg l^{-1} and 10.28 mg l^{-1} for the Bornholm Basin and the Baltic Proper, respectively. It should also be acknowledged that the upper confidence limit is an overestimate, since the standard deviation includes both natural interannual variation and uncertainty from determining annual values of oxygen debt. However, the variation around the trends lines (Fig. 3.1) has a similar magnitude over the entire period, despite considerably more data in recent years. This suggests that the uncertainty in determining annual values is relatively small compared to the interannual variations.

Since oxygen debt is calculated as the lack of oxygen from a fully saturated parcel of water, the effect of temperature increase on oxygen solubility over the study period is accounted for. However, this means that the limit, defining the upper range of the 'natural' oxygen debt distribution, cannot be directly translated to oxygen concentrations. Thus, there are two avenues for translating boundaries for oxygen debts to oxygen concentrations: 1) temperature increase will exacerbate hypoxia and thus the targets for oxygen should be adjusted to counter-compensate this; or 2) temperature increase is an exogenous pressure on the system, which should not affect the targets set for oxygen. The mean temperatures in the bottom waters before 1940 were approximately 5°C and 4.5°C for the Bornholm Basin and the Baltic Proper, respectively, increasing to more recent levels of 7°C and 5.5°C in the two basins (Fonselius & Valderama 2003). These temperature changes correspond to a lowering in oxygen solubility by 0.56 mg l^{-1} in the Bornholm Basin and 0.30 mg/l in the Baltic Proper. Following the first avenue, the natural limits for volume-specific debts would thus be converted into oxygen levels of 4.69 mg l^{-1} and 3.11 mg l^{-1} for the Bornholm Basin and the Baltic Proper, respectively. Compensating for the temperature increase experienced over the last 100 years would instead result in natural limits for oxygen levels of 5.25 mg l^{-1} and 3.41 mg l^{-1} for the two basins, respectively.

Defining targets for benthic invertebrates

Benthic invertebrates can be used for target setting according to the methodology outlined in Section 2.5. The challenge is to link this target to oxygen concentrations, assuming that this is the key driver, and thus to the planned recalculation of nutrient inputs. However, neither change point detection analyses nor attempts to establish a link between benthic invertebrates and oxygen concentrations have been made.

3.4 Summary of the target setting protocol

From the derivation of the proposed target in Chapter 2, it is evident that the different indicator targets should be valued with different confidences. Oxygen and Secchi depth have long-term trends covering a period where human disturbances are presumably low, and the targets have been defined as a significant deviation from the indicator distribution in a relatively unaffected

Table 3.14 Potential targets to be used for further work regarding the eutrophication segment of the Baltic Sea Action Plan. G1 = Group 1 targets; G2 = Group 2 targets; Chl-a = annual mean concentration of Chlorophyll a; TN = annual mean TN concentration; DIN = winter mean DIN concentration; TP = annual mean TP concentration; and DIP = winter mean DIP concentration. + = directly established target; (+) = indirectly established target; – = no target suggested.

Basin	G1		G2				
	Oxygen	Secchi	Chl-a	TN	DIN	TP	DIP
1. Northern Kattegat	–						
2. Central Kattegat	–	+	(+)	(+)	(+)	(+)	(+)
3. Southern Kattegat	–						
4. Samsø Belt	–						
5. Fehmarn Belt	–	+	(+)	(+)	(+)	(+)	(+)
6. The Sound	–						
7. Arkona Basin	–	+	(+)	(+)	(+)	(+)	(+)
8. Bornholm Basin	+	+	(+)	(+)	(+)	(+)	(+)
9. Baltic Proper	+	+	(+)	(+)	(+)	(+)	(+)
10. Bothnian Sea	–	+	(+)	(+)	(+)	(+)	(+)
11. Bothnian Bay	–	+	(+)	(+)	(+)	(+)	(+)
12. Gulf of Riga	–	+	(+)	(+)	(+)	(+)	(+)
13. Gulf of Finland	–	+	(+)	(+)	(+)	(+)	(+)

state. This approach is believed to give relatively high confidence in the targets.

The targets for nutrient and Chlorophyll *a* concentrations rest on weaker assumptions. First, there was considerable variation between the model results in the 1900 scenario, and consequently the combined estimates for the pre-eutrophication period are uncertain. Secondly, the same applies to the estimates from the early data periods, which are based on relatively few observations yielding relatively large uncertainty in the mean estimates. Finally, the targets were proposed as the average of two uncertain estimates (1900 and 1970s), since no other information is available to weigh these estimates against each other. Thus, we are less confident in the targets proposed for nutrients and Chl *a*.

Consequently, the basin-wise target setting is suggested to be made according to the following hierarchy: (1) Group 1 targets: Oxygen and Secchi depth, which are labelled directly and are observation-based targets; and (2) Group 2 targets: Chlorophyll *a* and nutrients, which are labelled indirectly and are preliminary established targets. It should be noted, however, that indicators for benthic invertebrates are not mature in the context of the BSAP mostly because of the link to pressures, for example oxygen concentrations are not yet quantitatively established. This hierarchy of targets with two distinct groups indicating the applicability of the derived targets at present is summarised in Table 3.14.

We suggest that the directly and observation-based targets from Group 1 are used for calculating the maximum allowable inputs in the revision of the BSAP, and for assessing whether an acceptable eutrophication status has been reached; and to use the indirectly and preliminary established targets from Group 2 as a safeguard that the ecological objectives for nutrients and algal blooms are not jeopardised in the sense that the calculated maximum allowable inputs should not lead to values for nutrients and Chl *a* exceeding the proposed targets. The nutrient and Chl *a* targets should also be seen as targets to be used in the assessment of the eutrophication status.

4. Conclusions

The trend and model analyses of indicators describing the five ecological objectives for eutrophication (nutrients, Chlorophyll *a*, water transparency, oxygen and benthic invertebrates) have clearly shown that all basins of the Baltic Sea have undergone significant changes over the last 100 years with decreasing Secchi depths and oxygen concentrations in tandem with increasing nutrient and Chlorophyll *a* concentrations.

The statistical trend analyses also suggest that this decline in ecosystem health has not been a continuous trend but rather a three-phase development from an early pre-eutrophication phase before ca. 1940; a eutrophication phase between 1940 and 1980; and a eutrophication stagnation phase after 1980, i.e. suggesting that the organic loading of the system has stabilised. Time series of Secchi depth and oxygen concentrations going back to the beginning of the 20th century document these different phases in the different basins. These three-phased trends are also consistent with trends in modelled nutrient inputs to the Baltic Sea showing a 3-5 factor increase from the early 1900s to the 1980s, followed by smaller declines in recent years.

However, it is also evident from the trend analyses of Secchi depth and oxygen that other factors have a significant perturbation on these variables. Secchi depth is affected by changes in other attenuating substances, not necessarily caused by nutrient enrichment, such as the enhanced input of coloured dissolved organic material (CDOM) from land. Oxygen conditions in bottom waters are affected by changes in the horizontal transport of dense saline water, vertical mixing across the halocline and temperature increases leading to reduced solubility of oxygen in water. Although there are no long-term trends for the concentration of CDOM, and given the mean levels and variations observed more recently, it has been assessed that changes in CDOM may only have significantly affected the Gulf of Finland and the Bothnian Bay. For the other basins, the levels and variations in CDOM are such that Secchi depths are only marginally affected. Long-term changes in bottom water salinity and stratification were combined with nutrient inputs and trends in oxygen debt, given as the loss of oxygen in a parcel of water that was originally saturated. The physical modulation of oxygen conditions was partially filtered out by means of a time series model, providing strong evidence for the direct anthropogenic influence on oxygen debt.

The evidence of a pre-eutrophication phase before ca. 1940 obtained from the analyses of both Secchi depth and oxygen debt suggests that in this period, the Baltic Sea was relatively unaffected by human activities, given that a considerable human population was inhabiting the catchment area. Particularly, the lack of a trend in oxygen debt in the Baltic Proper, where an excess organic loading of the bottom waters would lead to an increasing oxygen debt due to the long retention time leading to a cumulative effect, suggests that the Baltic Sea was capable of processing the relatively low

Table 4.1 *Proposed targets (Group 1 with higher confidence) for Secchi depth and oxygen debt in the Baltic Sea basins (summarised from Chapter 3). Oxygen debt targets (volume-specific for the deep water below the halocline) have been converted into oxygen concentration targets using the recent/historical temperature levels. For Secchi depth targets, the basins in the Kattegat as well as Samsø Belt, Fehmarn Belt and the Sound have been combined into a single unit. For oxygen targets, the Baltic Proper and the Gulf of Finland are considered as one common basin and the same target has been assigned to both basins. Summer means are June-September.*

Basin	Secchi depth target (m)		Oxygen debt target (mg l^{-1})	Oxygen concentration target (mg l^{-1})
	Summer	Annual		
1. Northern Kattegat	7.56	7.52		
2. Central Kattegat	7.56	7.52		
3. Southern Kattegat	7.56	7.52		
4. Northern Belt Sea	7.75	6.33		
5. Southern Belt Sea	7.75	6.33		
6. The Sound	7.75	6.33		
7. Arkona Basin	7.60	7.16		
8. Bornholm Basin	6.84	8.27	6.37	4.69 / 5.25**
9. Baltic Proper	7.55	8.78	8.66	3.11 / 3.41**
10. Bothnian Sea	6.79	6.89		
11. Bothnian Bay	6.31*	6.39*		
12. Gulf of Riga	3.74	3.90		
13. Gulf of Finland	5.42*	5.41*	8.66	3.11 / 3.41**

* Secchi depth targets may be lowered by up to 0.5 m if anticipated changes in CDOM properties are accounted for.
** Oxygen targets alleviated to compensate for temperature increases over the last 100 years.

nutrient inputs during this period. Consequently, this period was used to define the distributions of Secchi depth and oxygen debt in a relatively unaffected state, and by using these distributions define, on a statistical basis with 95% confidence, the boundaries for the natural variation within the period. Thus, the yearly means of Secchi depth below the 5-percentiles and oxygen debt exceeding the 95-percentiles of the distributions for these variables in the pre-eutrophication phase represent a significant deviation from a relatively unaffected situation. These percentiles (summarised in Table 4.1) are proposed as targets for the BSAP since they represent the boundary between a perceived relatively unaffected status and an affected status.

The proposed targets for Secchi depth and oxygen are affected by climate change. In the Bothnian Bay and the Gulf of Finland, the results suggest that up to 0.5 m of the decline in water transparency over the last 100 years could be caused by increasing levels of CDOM (see Section 2.3.2). If this shifting baseline caused by anticipated increasing CDOM concentrations is included in the target setting, the proposed targets for the Bothnian Bay and the Gulf of Finland can be reduced by a maximum of 0.5 m.

For the other basins, the potential effect of changing CDOM is small and cannot justify any adjustments to the targets.

Oxygen debt targets, which are independent of the temperature effect on oxygen solubility, can be converted into oxygen concentration targets, but this back-calculation depends on whether oxygen solubility is calculated from recent temperatures or temperatures at the beginning of the 20th century. Thus, as temperature increases over the last 100 years have reduced the natural oxygen supply to the bottom waters, it should be decided if this temperature change should lead to a tightening of targets, i.e. if nutrient inputs should be further reduced to counteract the effects of temperature increases. If the oxygen targets should also include the effect of temperature increases, then the targets for oxygen concentration should be increased by 0.56 and 0.30 mg l^{-1} in the Bornholm Basin and the Baltic Proper, respectively.

Trends for nutrients and Chlorophyll *a*, the latter being a proxy indicator for the algal bloom objective, only cover the last 4-5 decades and thus part of the eutrophication phase and the eutrophication stagnation phase; however, there are no monitoring data to assess the potential level for these variables in the pre-eutrophication phase. Such estimates, albeit quite variable across models, have been produced with the three simulation models; moreover, the targets are proposed as the average of the model predictions for the period around 1900 and the levels estimated from the earliest data period in the 1970s. This approach was not possible for TN and TP since these variables are not included in the simulation models, and for Chl *a* because the simulation results were not considered reliable. Instead, the estimated levels for TN, TP and Chl *a* in the 1970s are suggested as an upper boundary for the targets that should not be compromised, realising that the Baltic Sea was already affected by eutrophication in the 1970s. The proposed targets for nutrients and Chl *a* are summarised in Table 4.2.

Some caution should be raised towards the use of DIN, DIP, and Chl *a* in the current target setting. These indicators are highly dependent upon changes in the seasonal pattern, particularly the length of the productive period. If the productive period has been extended as a result from climate change, the spring bloom may develop earlier, leading to an earlier depletion of inorganic nutrients and overall higher annual mean for Chl *a*. It is similarly likely that production may extend longer into the autumn period due to later development of ice cover and break-down of the thermocline, which will lead to a reduced accumulation of inorganic nutrients during winter. Such shifts in the productive period may have strong consequences for the DIN, DIP and Chl *a* indicators, but are not expressions of eutrophication. TN and TP indicators are more robust, albeit not entirely, to such changes, since they are independent of the internal transformations between the inorganic and organic pools. Therefore, fulfilment of the nutrient ecological objective should mainly be based on targets for total nutrients.

For nutrients, Chl *a* and Secchi depth targets for both annual and seasonal means are given. In general, indicators based on annual means where the seasonal variation is properly accounted for have better precision because they are based on more data. It is therefore recommended to use annual means from a statistical point of view. There can be practical limitations, such as ice cover in the Bothnian Bay that do not allow for estimating annual means; in such cases, seasonal means should be preferred as the indicator. Finally, the ecological relevance of seasonal versus annual means is also an important aspect.

The proposed targets for Secchi depths, nutrients and Chl *a* have been combined with the spatial models (Figs. 2.2, 2.5, 2.8, 2.11, 2.14 and 2.17) to calculate targets for the HELCOM sub-divisions (Annex F). These targets are generally consistent with those calculated for the BALTSEM basins (Tables 4.1 and 4.2), although there can be some smaller changes due to differences in the spatial delineation of the BALTSEM and HELCOM divisions.

In summary, the analyses in the present report have led to a three-level grouping of the indicator

Table 4.2 *Proposed targets (Group 2 with lower confidence) for nutrients (in µmol l^{-1}) and Chlorophyll a (in µg l^{-1}) in the Baltic Sea basins (summarised from Chapter 3). For TN, TP and Chl a, specific targets are not given, but it is recommended to use targets below the suggested values. Winter means are December-February and summer means are June-September.*

Basin	Winter		Summer	Annual				
	DIN	DIP	Chl α	DIN	DIP	TN	TP	Chl α
1-3 Kattegat	4.07	0.49	<1.22	1.52	0.21	<17.43	<0.64	<1.45
4-6 Danish Straits	5.11	0.58	<1.89	1.70	0.32	<21.79	<0.97	<1.79
7. Arkona Basin	2.62	0.38	<1.44	1.03	0.22	<17.36	<0.66	<1.36
8. Bornholm Basin	2.70	0.31	<2.44	1.14	0.19	<16.29	<0.57	<1.20
9. Baltic Proper	2.62	0.29	<1.74	1.45	0.18	<16.23	<0.44	<0.93
10. Bothnian Sea	2.75	0.19	<1.52	1.57	0.13	<15.66	<0.24	<1.33
11. Bothnian Bay	5.08	0.07	<1.63	4.19	0.07	<16.88	<0.18	<1.23
12. Gulf of Riga	<6.80	<0.41	<4.12	<4.21	<0.26	<37.98	<0.71	<5.10
13. Gulf of Finland	6.79	0.65	<4.37	3.35	0.27	<22.15	<0.56	<2.54

targets for the five ecological objectives based on a confidence rating of the targets: 1) directly and observation-based targets for water transparency and oxygen; 2) indirectly and preliminary established targets for nutrients and algal blooms; and 3) pre-mature targets for benthic invertebrate communities. Particularly, there is a need to establish quantitative causal links between the indicators of the benthic community status and pressures such as oxygen. Based on the confidence applied to the different targets, it is suggested that Group 1 targets with higher confidence are employed as absolute targets that should not be compromised; and that Group 2 targets with lower confidence are employed as guiding targets aimed to be fulfilled. If the proposed targets are met with sufficient confidence, the status of the Baltic Sea will likely resemble the situation before 1940, which is believed to be an ecosystem relatively unaffected by eutrophication.

4.1 Comparing the targets with the present status and BSAP reductions

The proposed targets above are compared to the present status (1997-2006) of the indicators obtained from the trend analyses as well as the predictions from the BALTSEM model for the baseline scenario (1997-2006) and the BSAP reduction scenario (Table 2.3). The output from BALTSEM was used since this will be the model employed to calculate the maximum allowable inputs in the revision of the BSAP. TN and TP cannot be computed with BALTSEM, while Chl *a* is not used because of the large uncertainty associated with translating the phytoplankton biomass of the different algae groups into Chl *a* (see above).

The proposed targets for Secchi depth are fulfilled only for annual Secchi depth (status) in the Danish Straits and the Arkona Basin (Table 4.3), although the proposed targets are close to being fulfilled for some of the other basins. The BALTSEM estimates for the baseline are mostly higher than the indicator status, except for the Danish straits, and the Arkona and Bornholm Basins, suggesting that there could be some bias between the targets and model predictions. The reduction scenario fulfils the proposed Secchi depth targets in some, but not all basins.

Within TARGREV, as the models were not customised to calculate the oxygen debt indicator, the proposed targets for oxygen could not be compared with the model output from BALTSEM. However, during the baseline the status of the oxygen debts in the Bornholm Basin and the Baltic Proper were 6.98 and 9.70 mg l^{-1}, respectively, and thus 0.60 and 1.03 mg l^{-1} above the targets.

The current status of winter DIN is generally above the proposed targets, whereas the current status of annual DIN is generally closer to the proposed targets (Table 4.4). This apparent lack of consistency between the two sets of target and status could be due to a shift in seasonality over time that will, in turn, affect the annual DIN means stronger than winter DIN means, stress-

Table 4.3 *Comparison of the proposed targets for Secchi depth (Table 4.1; in m) with the indicator status level during the baseline (Table E.6, 1997-2006), the BALTSEM predictions for the baseline and reduction scenarios (RED) (cf. Table 2.3).*

Basin	Summer (Jun-Sep)				Annual			
	Target	Status	Baseline	RED	Target	Status	Baseline	RED
1-3 Kattegat	7.56	7.48	7.82	8.37	7.52	6.60	8.48	8.81
4-6 Danish Straits	7.75	6.78	6.50	7.31	6.33	7.19	7.59	8.04
7. Arkona Basin	7.60	7.33	5.79	6.66	7.36	8.20	6.70	7.16
8. Bornholm Basin	6.84	6.59	5.67	6.53	8.05	7.93	6.45	6.95
9. Baltic Proper	7.55	5.72	5.87	6.51	8.78	7.43	6.57	6.89
10. Bothnian Sea	6.79	4.93	5.63	5.79	6.89	4.95	5.97	6.05
11. Bothnian Bay	6.31*	4.48	4.71	4.72	6.39*	4.70	4.85	4.85
12. Gulf of Riga	3.74	3.28	4.90	5.31	3.90	3.30	5.58	5.80
13. Gulf of Finland	5.42*	3.27	4.61	5.08	5.41*	3.73	5.57	5.80

** Secchi depth targets may be lowered by up to 0.5 m if anticipated changes in CDOM properties are accounted for.*

Table 4.4 *Comparison of the proposed targets for DIN (Table 4.2; in µmol l[-1]) with the indicator status level during the baseline (Table E.2, 1997-2006), the BALTSEM predictions for the baseline and reduction scenarios (RED) (cf. Table 2.3).*

Basin	Winter (Dec-Feb)				Annual			
	Target	Status	Baseline	RED	Target	Status	Baseline	RED
1-3 Kattegat	4.07	6.27	4.95	4.56	1.52	1.83	1.65	1.52
4-6 Danish Straits	5.11	7.27	4.38	3.88	1.70	2.38	1.51	1.33
7. Arkona Basin	2.62	5.03	3.02	2.76	1.03	1.34	1.25	1.14
8. Bornholm Basin	2.70	3.70	3.42	3.02	1.14	1.35	1.50	1.37
9. Baltic Proper	2.62	3.68	4.31	3.49	1.45	1.70	2.21	1.75
10. Bothnian Sea	2.75	3.60	4.22	4.56	1.57	1.98	2.58	3.31
11. Bothnian Bay	5.08	7.35	6.29	6.98	4.19	5.05	5.94	6.65
12. Gulf of Riga	<6.80	9.00	5.92	6.62	<4.21	5.59	3.41	3.81
13. Gulf of Finland	6.79	10.37	6.51	6.39	3.35	3.68	4.16	4.04

ing the point that DIN is not a robust indicator in the face of climate change. There were also some differences between the observed and modelled baseline status, suggesting that there could be some bias between the BALTSEM model and the indicators.

The current status of winter DIP is close to the targets in the Danish Straits and Kattegat as well as in the Gulf of Bothnia, whereas the status is higher in the central parts of the Baltic Sea (Table 4.5). The same pattern is partly reflected in the annual DIP means, except that the DIP status is below the target in the Gulf of Riga. However, as stressed above for DIN, DIP means are also sensitive to changes in the seasonal pattern caused by climate change, thus making it more difficult to compare both winter and annual means across decades. There are also some differences between the observed and modelled DIP levels, suggesting that there could be some bias between the BALTSEM model and the indicators.

The current levels of TN are all above the upper boundary for the TN targets with the exception of the Danish Straits (Table 4.6). It also appears from the comparison with the Kattegat and the Arkona Basin that the upper target boundary in the Danish Straits could be too high. The current levels of TP are below the upper boundary for the targets in the Kattegat and Danish Straits as well as the Bothnian Bay (Table 4.6).

The current levels of summer Chl *a* are above the upper boundary for the targets, except for the Bornholm Basin (Table 4.6). However, the upper boundary in the Bornholm Basin appears high when compared to the levels in the surrounding basins (Arkona Basin and Baltic Proper). For the annual Chl *a* means, the current status is above the

Table 4.5 *Comparison of the proposed targets for DIP (Table 4.2; in µmol l[-1]) with the indicator status level during the baseline (Table E.4, 1997-2006), the BALTSEM predictions for the baseline and reduction scenarios (RED) (cf. Table 2.3).*

Basin	Winter (Dec-Feb)				Annual			
	Target	Status	Baseline	RED	Target	Status	Baseline	RED
1-3 Kattegat	0.49	0.49	0.61	0.55	0.21	0.15	0.29	0.25
4-6 Danish Straits	0.58	0.60	0.63	0.50	0.32	0.23	0.34	0.24
7. Arkona Basin	0.38	0.52	0.50	0.30	0.22	0.27	0.33	0.16
8. Bornholm Basin	0.31	0.56	0.55	0.33	0.19	0.32	0.36	0.18
9. Baltic Proper	0.29	0.49	0.80	0.46	0.18	0.21	0.51	0.27
10. Bothnian Sea	0.19	0.20	0.49	0.33	0.13	0.09	0.33	0.21
11. Bothnian Bay	0.07	0.05	0.12	0.10	0.07	0.05	0.08	0.07
12. Gulf of Riga	<0.41	0.78	1.08	0.83	<0.26	0.20	0.73	0.51
13. Gulf of Finland	0.65	1.01	1.09	0.76	0.27	0.36	0.77	0.50

Table 4.6 *Comparison of the proposed targets for TN and TP (Table 4.2; in μmol l^{-1}) as well as Chl a (Table 4.2; in μg l^{-1}) with the indicator status level during the baseline (Tables E.1, E.3 and E.5, 1997-2006). Summer means are June-September.*

Basin	Annual TN		Annual TP		Summer Chl *a*		Annual Chl *a*	
	Target	Status	Target	Status	Target	Status	Target	Status
1-3 Kattegat	<17.43	18.50	<0.64	0.57	<1.22	1.35	<1.45	1.75
4-6 Danish Straits	<21.79	20.59	<0.97	0.71	<1.89	2.32	<1.79	2.17
7. Arkona Basin	<17.36	20.95	<0.66	0.70	<1.44	1.93	<1.36	1.70
8. Bornholm Basin	<16.29	21.50	<0.57	0.70	<2.44	2.24	<1.20	1.97
9. Baltic Proper	<16.23	20.78	<0.44	0.59	<1.74	2.79	<0.93	1.84
10. Bothnian Sea	<15.66	17.13	<0.24	0.32	<1.52	2.38	<1.33	2.08
11. Bothnian Bay	<16.88	19.16	<0.18	0.17	<1.63	2.26	<1.23	1.60
12. Gulf of Riga	<37.98	29.31	<0.71	0.84	<4.12	4.26	<5.10	4.47
13. Gulf of Finland	<22.15	23.75	<0.56	0.87	<4.37	4.98	<2.54	3.31

upper boundary for the targets in all basins, except for the Gulf of Riga where the target was based on data from 1990-95 and thus should be considerably lower than the indicated upper boundary.

In summary, when comparing the current status to the proposed target levels, acknowledging a belief that the current status does not fulfil the ecological objectives, the proposed targets for Secchi depth, oxygen, TN, TP and Chl *a* appear robust, whereas the targets based on historical levels of DIN and DIP and assessed by recent levels are sensitive to changing seasonality in the uptake of nutrients for primary production and are thus not robust to climate change. Accordingly, we suggest that DIN and DIP are given less weight as indicators of nutrient status but rather focus more on TN and TP. Finally, the comparisons indicate that biases between the observed and modelled (using BALTSEM) indicators could be present, and any such potential bias should be resolved for the calculation of the maximum allowable inputs.

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ANNEX A: Target setting concepts and their definitions

The concept of 'reference conditions' in the context of European water policies is derived from the Water Framework Directive (Anon. 2000).

The WFD definition reads:
Reference condition (noun) - is a description of the biological quality elements that exist, or would exist, at high status, that is, with no, or very minor disturbance from human activities.

The objective of setting reference condition standards is to enable the assessment of ecological quality against these standards.

The concpt of reference condition has a number of strengths:

- It is a well consolidated concept used in all EU coastal and transitional waters – mostly because it originates from the WFD.
- Reference conditions are generally determined by scientific methods, e.g. the analysis of historical data, modelling (hindcast scenarios) and reference sites – or by a combination of these three methods, the latter sometimes referred to as expert judgement.
- Over the last decade, the amount of scientific literature on reference conditions has increased significantly
- In principle, scientifically sound reference conditions will result in more accurate target setting.

It should also be emphasised that HELCOM's integrated thematic assessment of eutrophication, in particular the classification of eutrophication status (HELCOM 2009), is based on basin-, site- or water-body-specific information on reference conditions.

Some weaknesses of the concept are:

- The current use of reference conditions within the Baltic Sea is not 100% harmonised - it is possible to identify a small number of site-specific 'outliers' when reference conditions are compared along a salinity gradient (see HELCOM 2006).
- There are no undisturbed reference sites in the Baltic Sea, and hence one method less for setting reference conditions is available; and partly due to this, the previous approaches (e.g. HELCOM 2006, 2009) relied to a rather high extent on expert judgement.
- Setting inaccurate values of reference conditions will result in inaccurate target values.

The concept of 'acceptable deviation' from reference conditions also originates from the Water Framework Directive, where the acceptable deviations are no deviation or a slight deviation from the reference conditions, whilst deviations being 'moderate' or 'high' are regarded 'unacceptable deviations' indicative of impaired ecological status.

The term 'acceptable' is by www.yourdictionary.com defined as:
Acceptable (adjective) – worth accepting; satisfactory, merely adequate; adequate to satisfy a need, requirement, or standard; or satisfactory.

The term 'deviation' is by www.yourdictionary.com defined as:
Deviation (noun) – the act or an instance of deviating; specifically: 1) sharp divergence from normal behaviour; 2) divergence from the official ideology or policies of a political party, esp. a Communist party; 3) the deflection of a magnetic compass needle due to magnetic influence; and 4) statistics the amount by which a number differs from an average or other comparable value.

A combination of the above two terms would lead to the following interpretation of Acceptable Deviation:
A divergence worth accepting or a divergence within a range considered normal.

The objective of setting an acceptable deviation from reference conditions is to define the boundary between acceptable and unacceptable eutrophication status and thus the setting of operational targets for relevant eutrophication indicators.

ANNEX B: Basin-specific trends and seasonal variations for nutrients, Chl *a* and Secchi depth

This annex contains figures similar to those presented for the Baltic Proper in Section 2 for the other basins in the Baltic Sea. These were omitted from the main report to save space.

Total nitrogen (TN)

Long-term temporal variation in TN concentrations in different basins (μmol l^{-1})

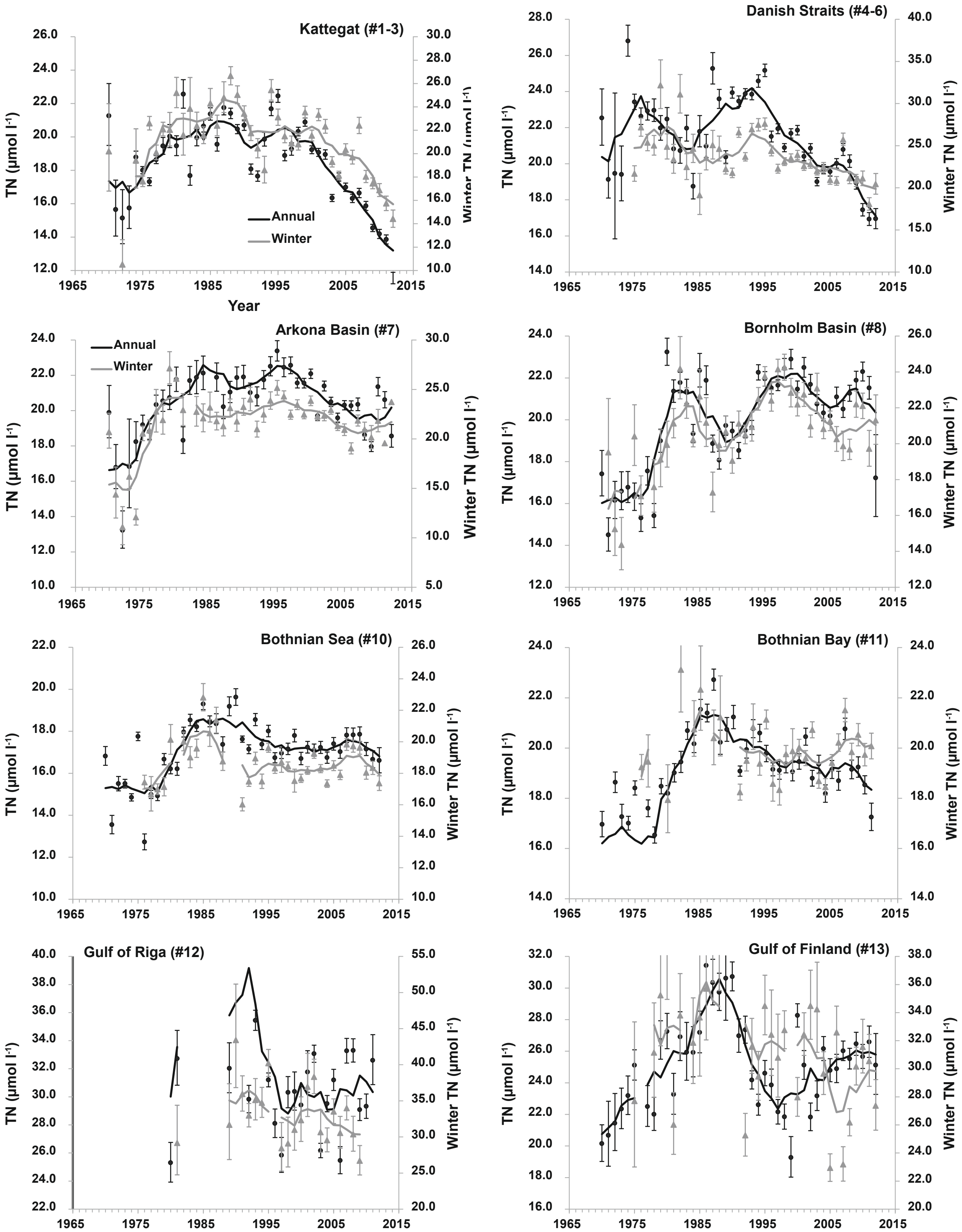


Figure B.1 *Long-term trend in annual (black) and winter (grey) surface TN concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means.*

Seasonal variation in TN concentrations in different basins (µmol l⁻¹)

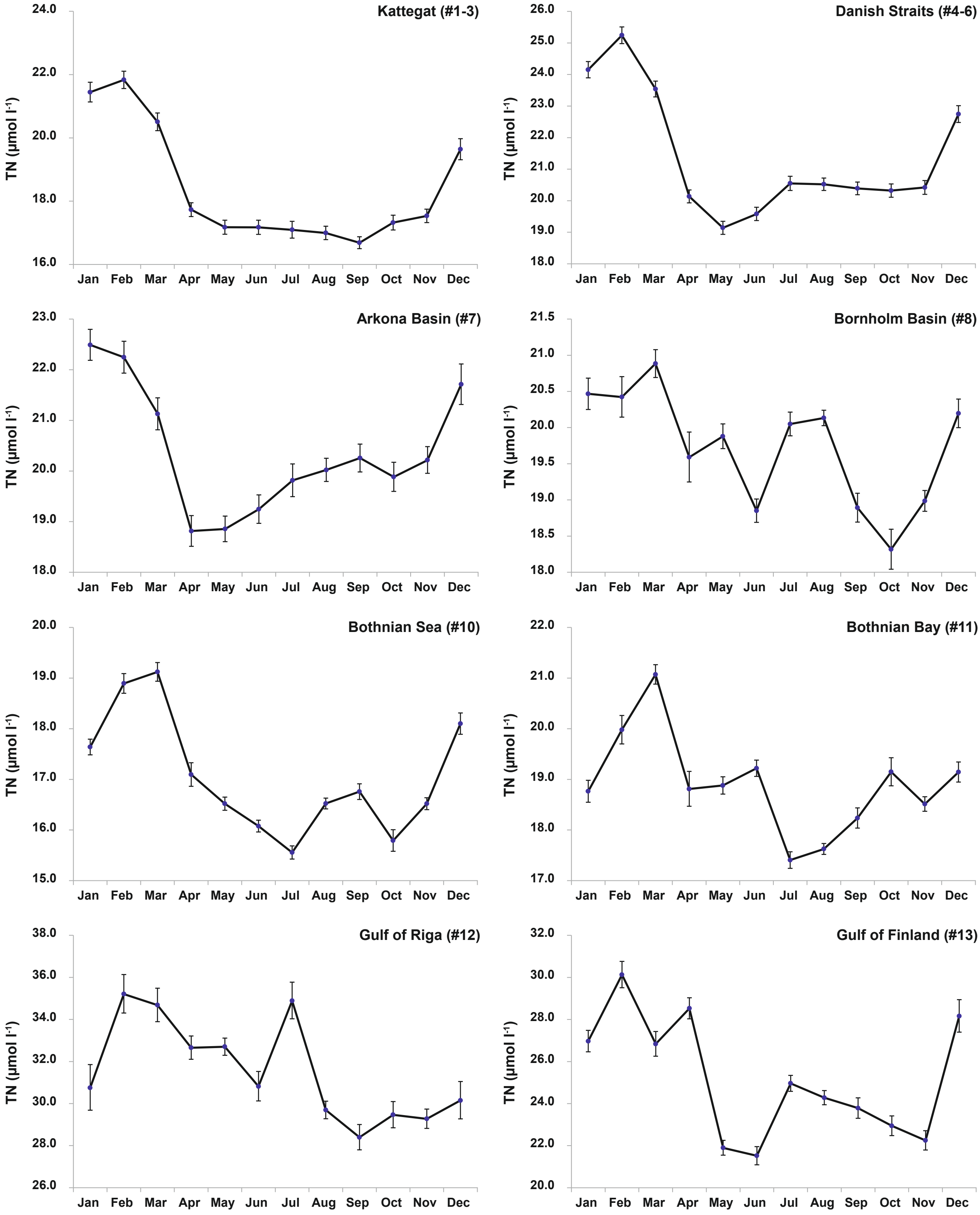


Figure B.2 *Seasonal variation in the mean surface TN concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Error bars represent 95% confidence limits of the means. Periods covered in the seasonal are given in the trends plots (Fig. B.1).*

Dissolved inorganic nitrogen (DIN)

Long-term temporal variation in DIN concentrations in different basins (μmol l^{-1})

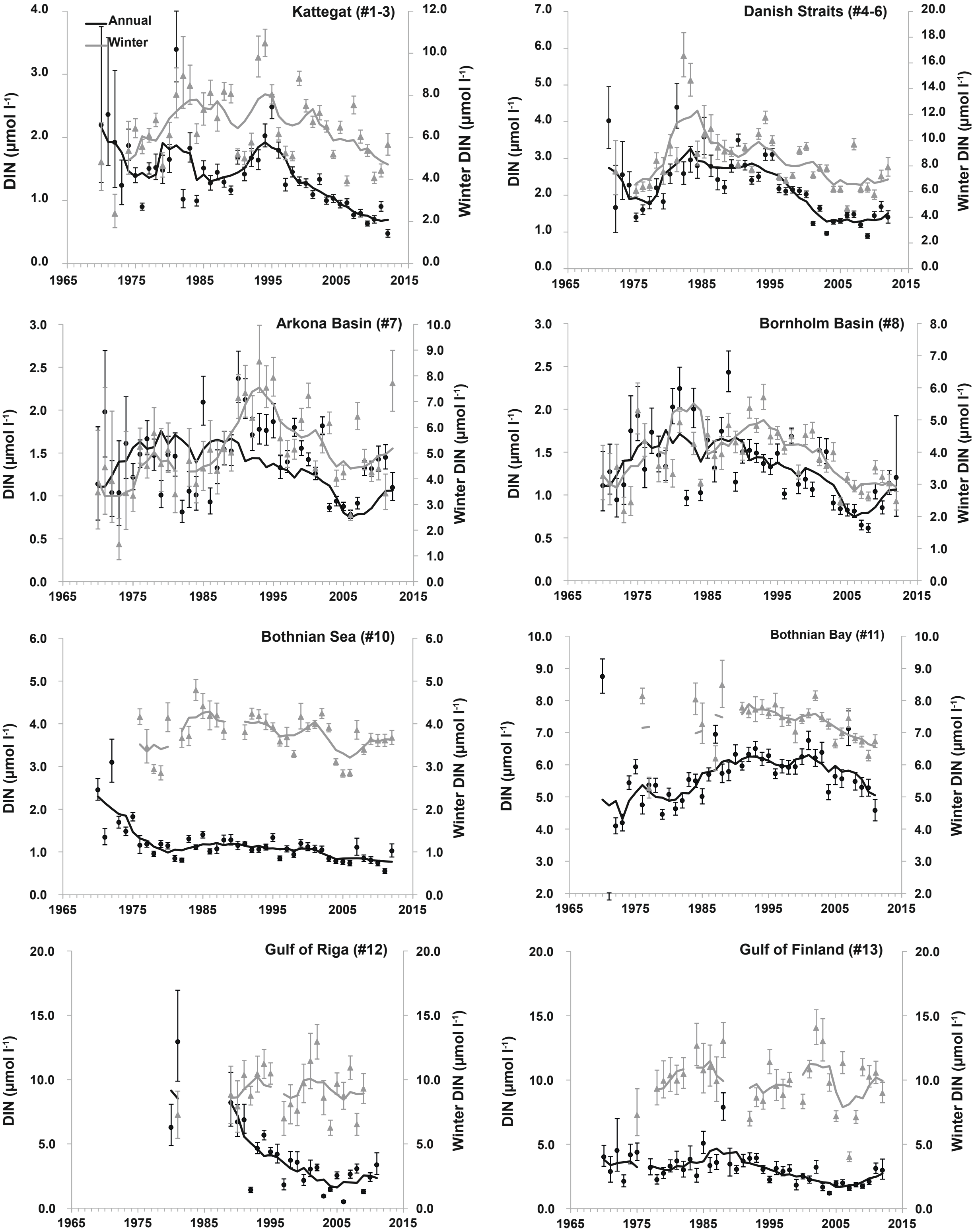


Figure B.3 *Long-term trend in annual (black) and winter (grey) surface DIN concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means.*

Seasonal variation in DIN concentrations in different basins (µmol l⁻¹)

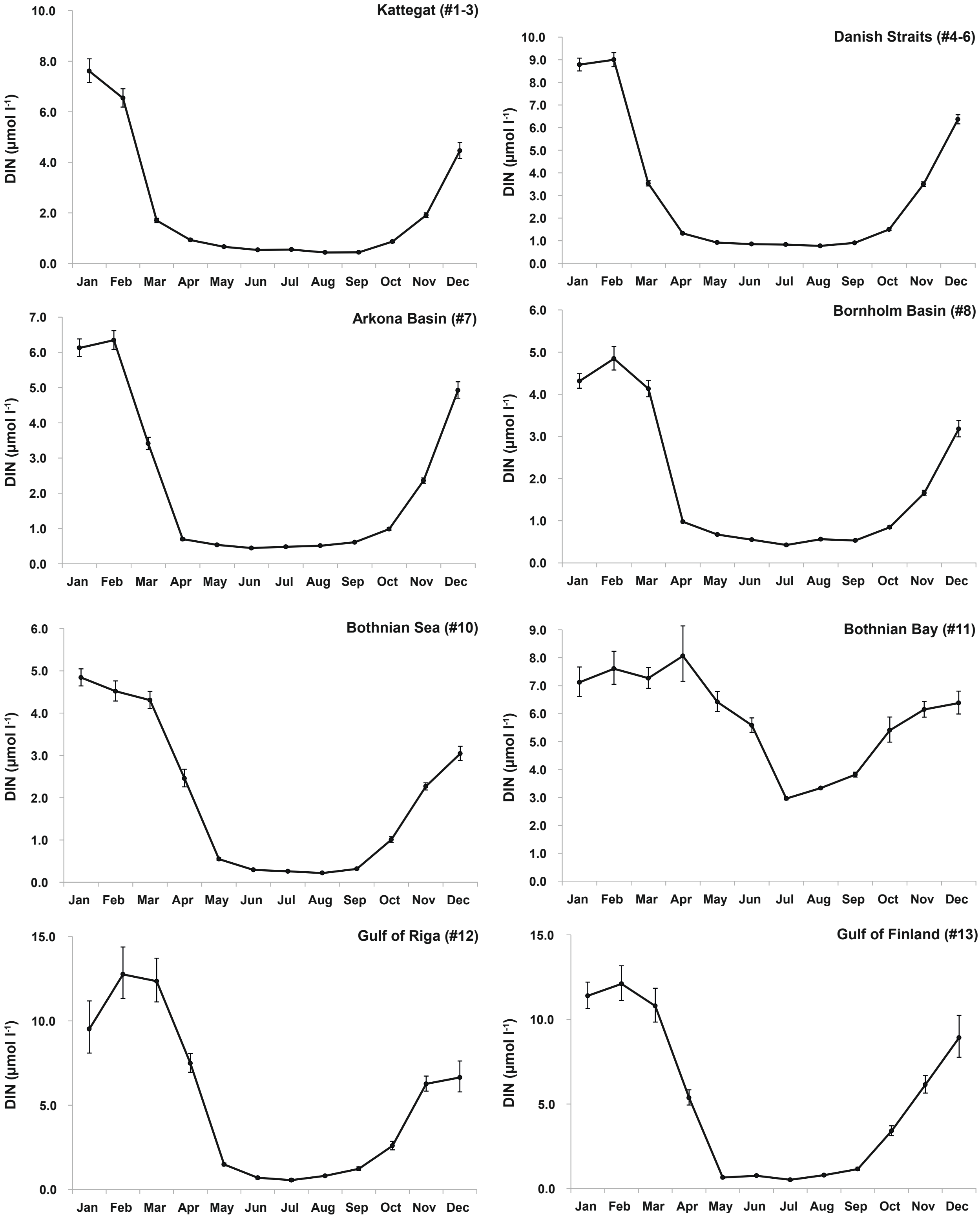


Figure B.4 *Seasonal variation in the mean surface DIN concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Error bars represent 95% confidence limits of the means. Periods covered in the seasonal are given in the trends plots (Fig. B.3).*

Total phosphorus (TP)

Long-term temporal variation in TP concentrations in different basins (µmol l-1)

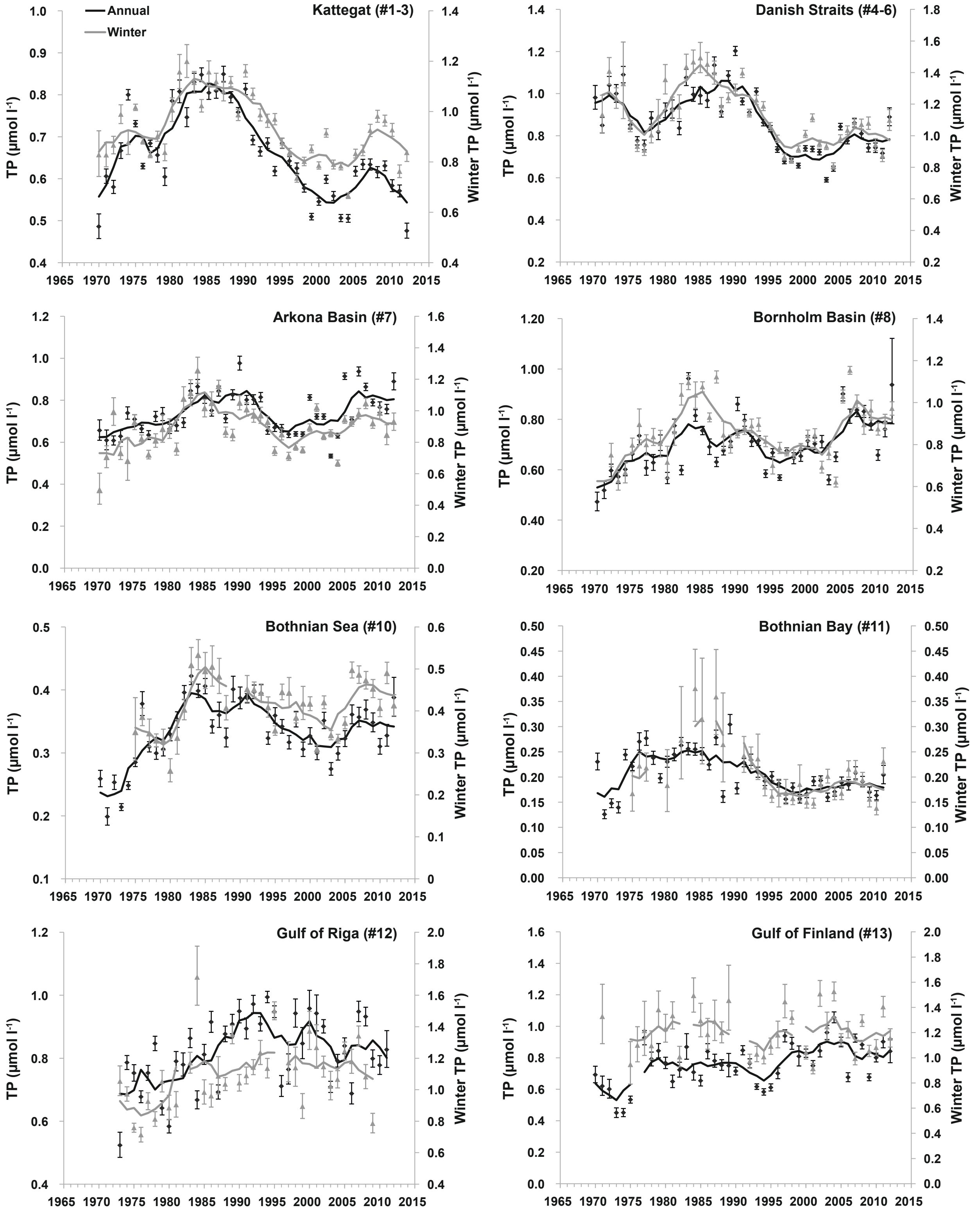


Figure B.5 *Long-term trend in annual (black) and winter (grey) surface TP concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means.*

Seasonal variation in TP concentrations in different basins (µmol l^{-1})

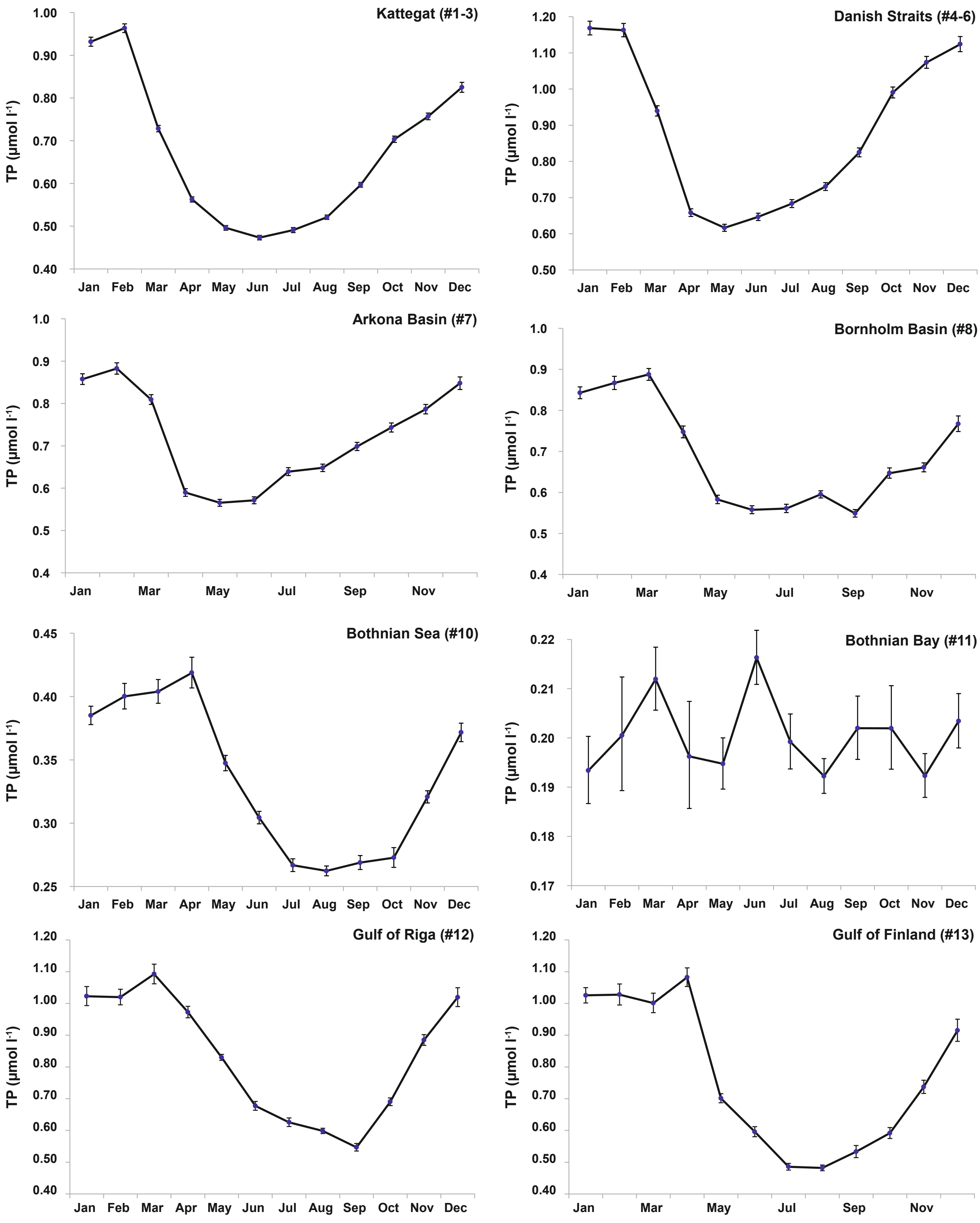


Figure B.6 *Seasonal variation in the mean surface TP concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Error bars represent 95% confidence limits of the means. Periods covered in the seasonal are given in the trends plots (Fig. B.5).*

Dissolved inorganic phosphorus (DIP)

Long-term temporal variation in DIP concentrations in different basins (μmol l⁻¹)

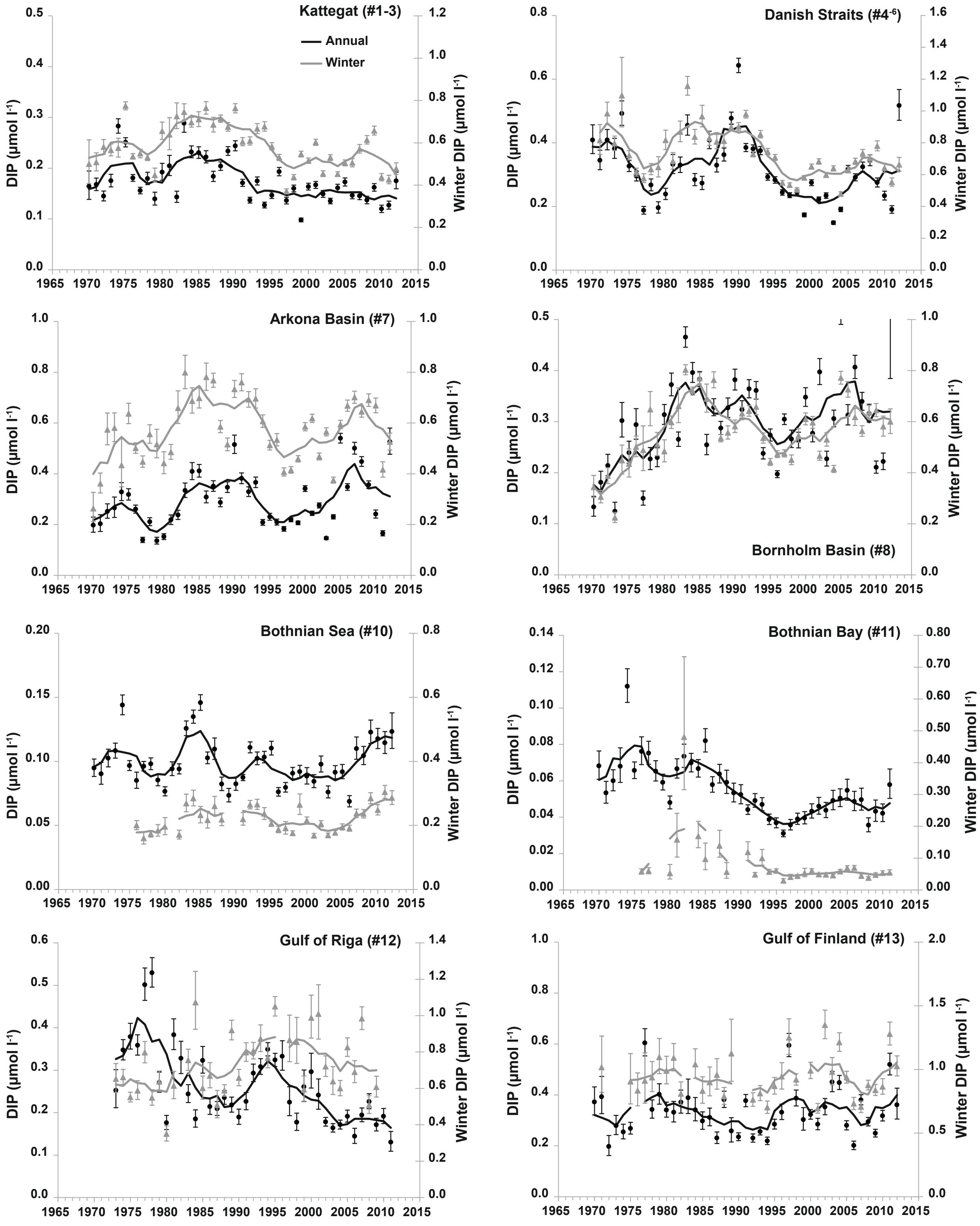


Figure B.7 *Long-term trend in annual (black) and winter (grey) surface DIP concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Lines indicate the five-year moving average (starting from 1970) and error bars represent 95% confidence limits of the means.*

Seasonal variation in DIP concentrations in different basins (μmol l^{-1})

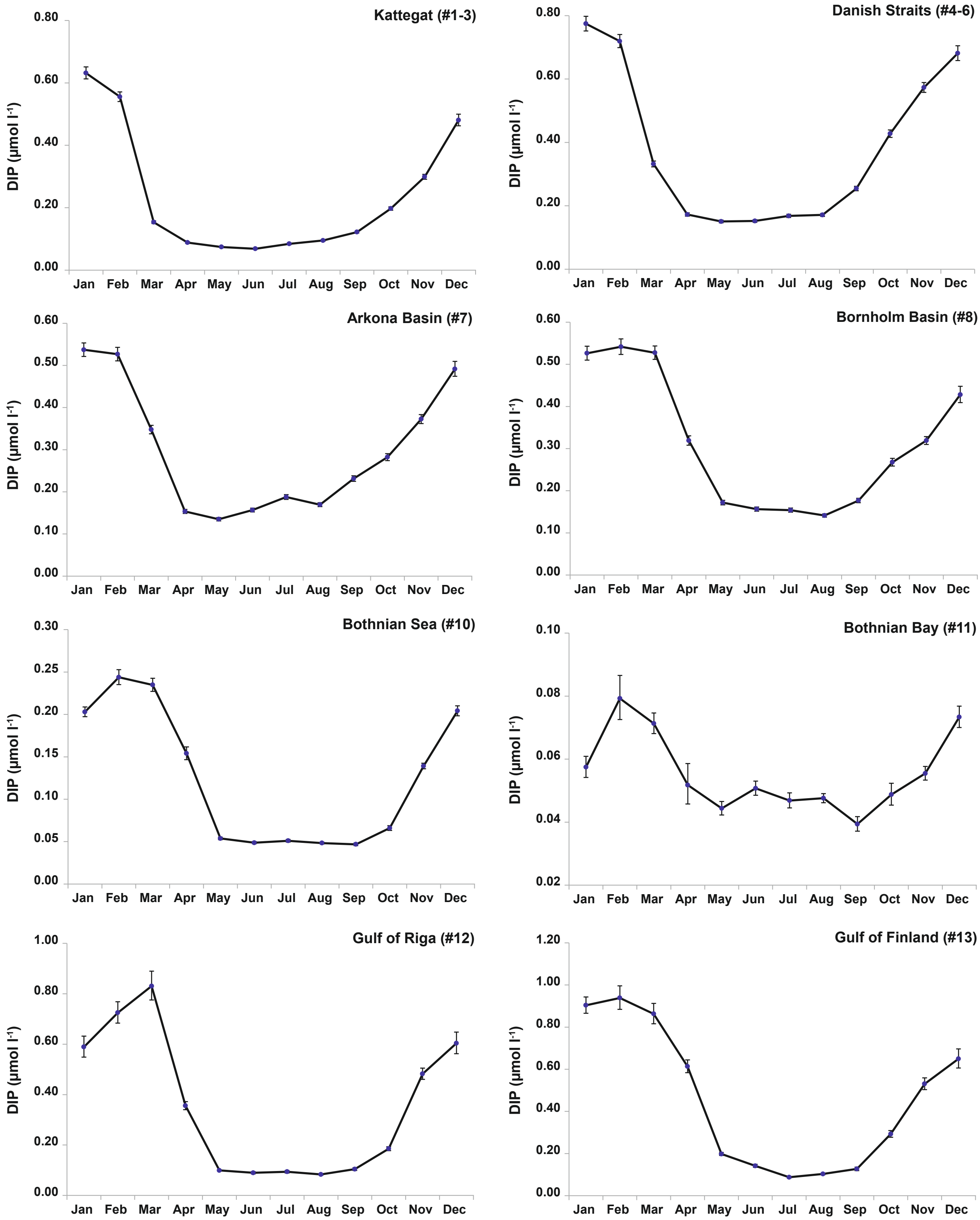


Figure B.8 *Seasonal variation in the mean surface TP concentrations in the different basins (0-10 m for Kattegat and Danish Straits; 0-20 m for others). Error bars represent 95% confidence limits of the means. Periods covered in the seasonal are given in the trends plots (Fig. B.7).*

Chlorophyll a

Long-term temporal variation in Chl a concentrations in different basins (µg l[-1])

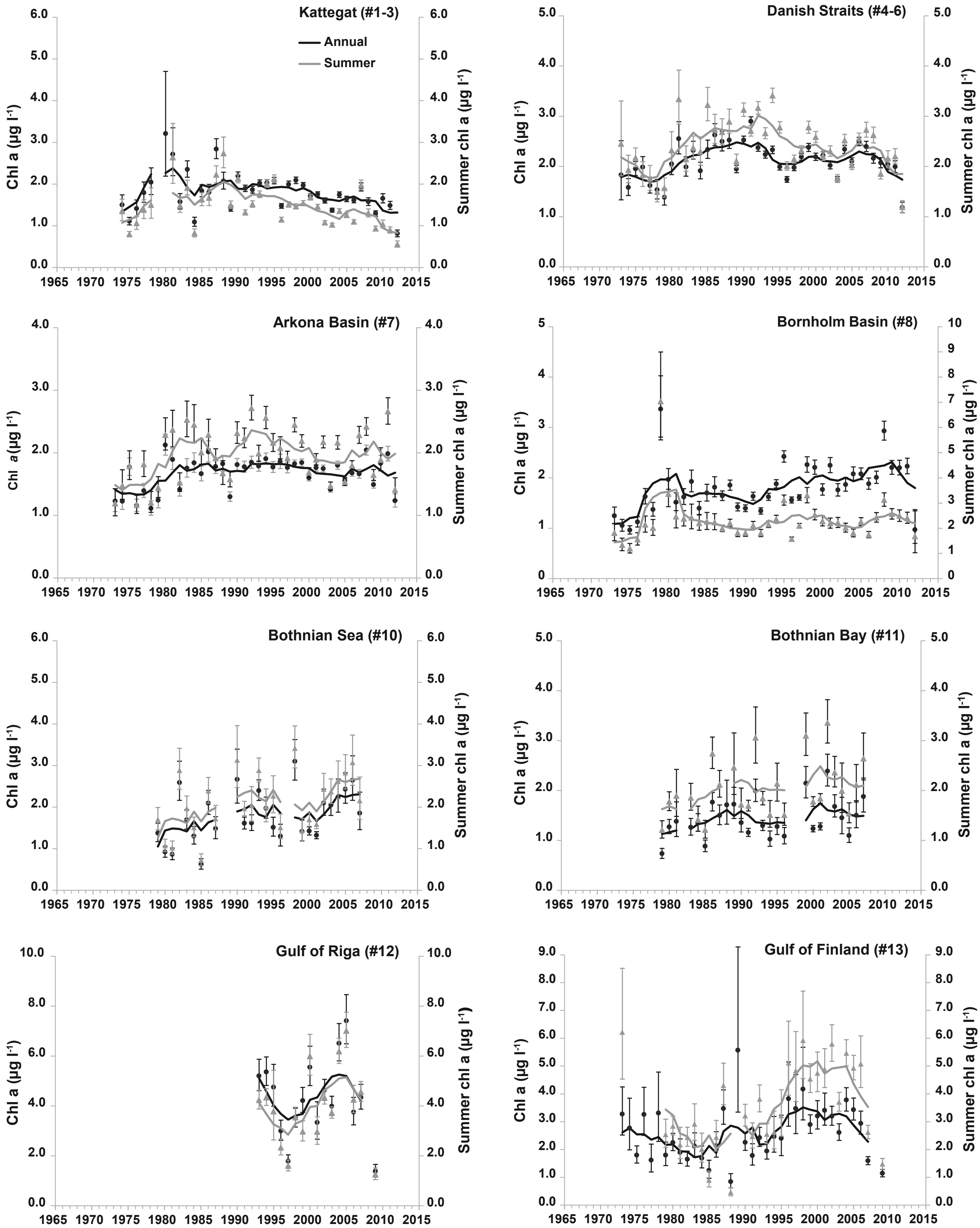


Figure B.9 *Long-term trend in annual (black) and summer (grey) surface Chl a concentrations in the different basins (0-10 m). Lines indicate the five-year moving average and error bars represent 95% confidence limits of the means.*

Seasonal variation in Chl a concentrations in different basins ($\mu g\ l^{-1}$)

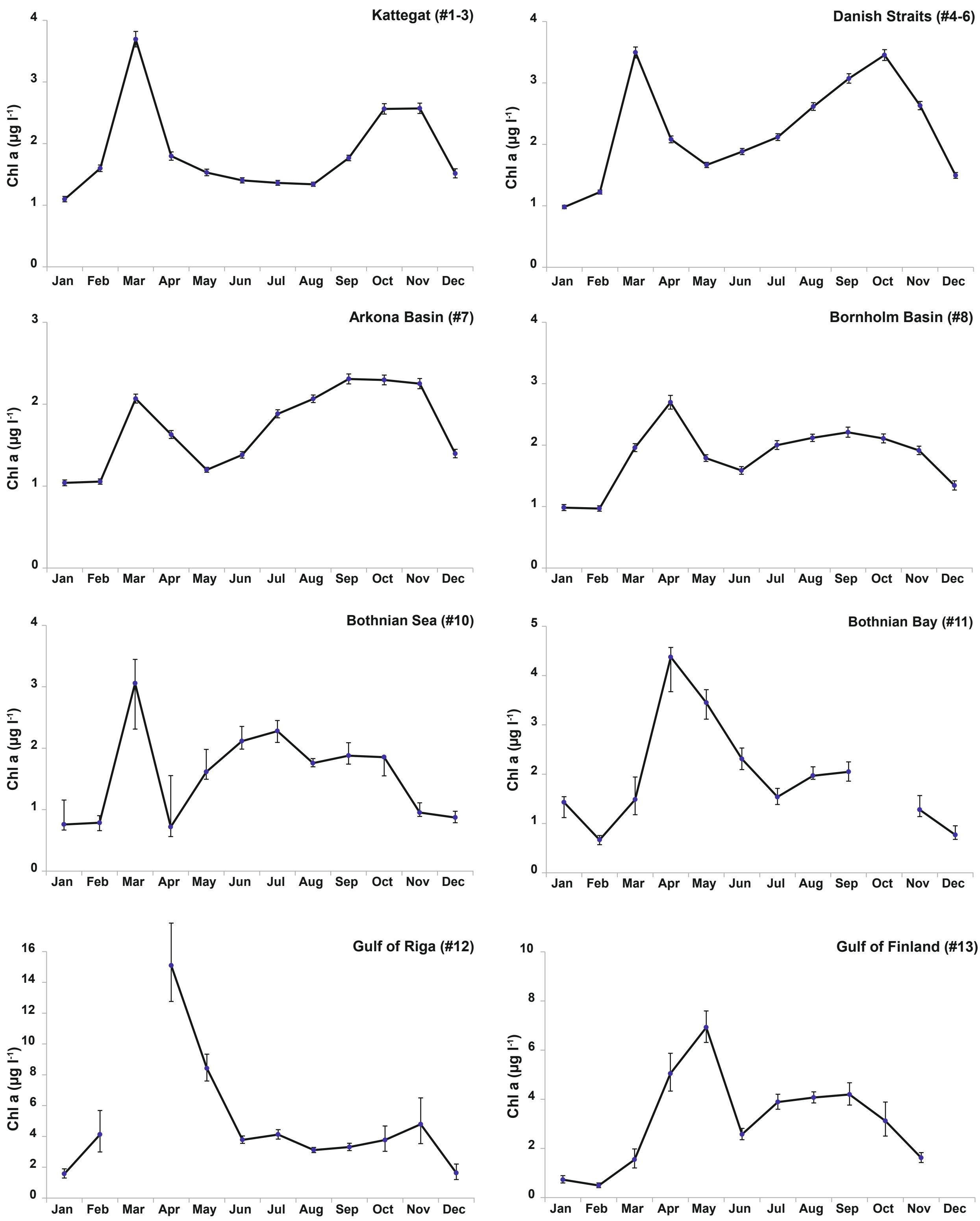


Figure B.10 *Seasonal variation in the mean surface Chl a concentrations in the different basins (0-10 m). Error bars represent 95% confidence limits of the means. Periods covered in the seasonal are given in the trends plots (Fig. B.9).*

Secchi depth

Long-term temporal variation in Secchi depths in different basins (m)

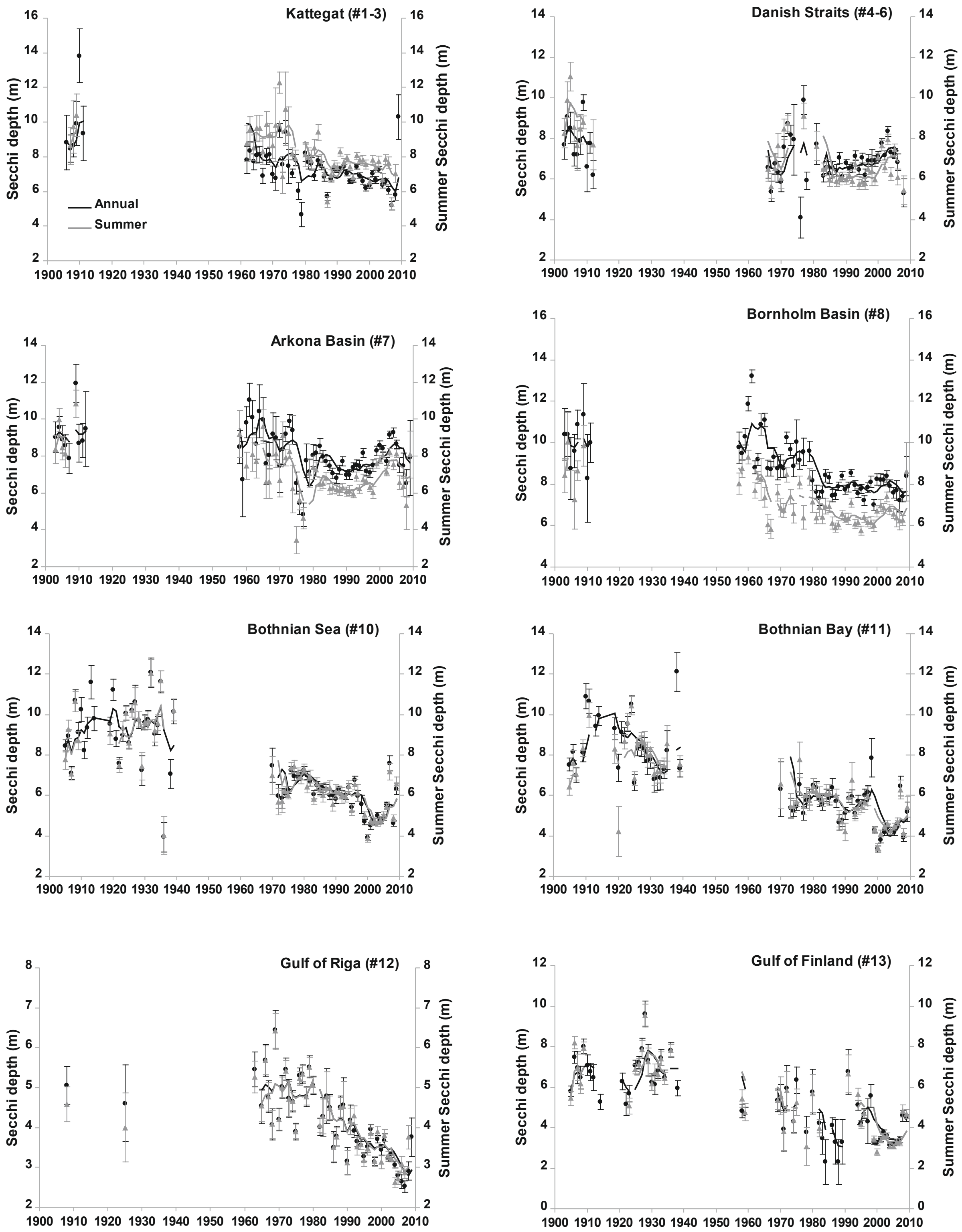


Figure B.11 *Long-term trend in annual (black) and summer (grey) Secchi depth in the different basins. Lines indicate the five-year moving average and error bars represent 95% confidence limits of the means.*

Seasonal variation in Secchi depths in different basins (m)

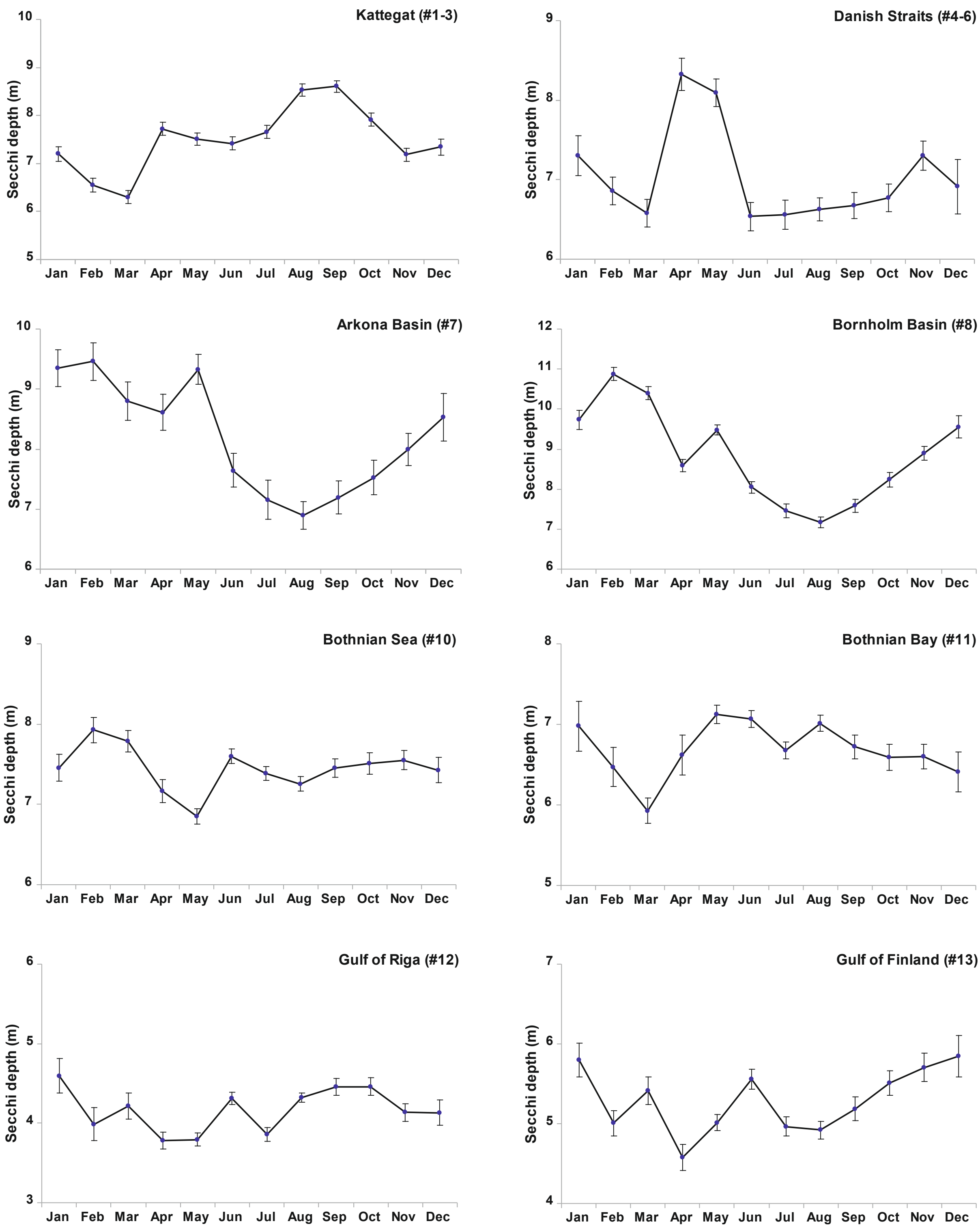


Figure B.12 *Seasonal variation in the mean Secchi depth in the different basins. Error bars represent 95% confidence limits of the means. Periods covered in the seasonal are given in the trends plots (Fig. B.11).*

Annex C: Trends and seasonality in estimates from salinity and oxygen profiles

As there is no permanent halocline in the Gulf of Riga, only the estimates of surface salinity are given.

Salinity

Trends in surface salinity

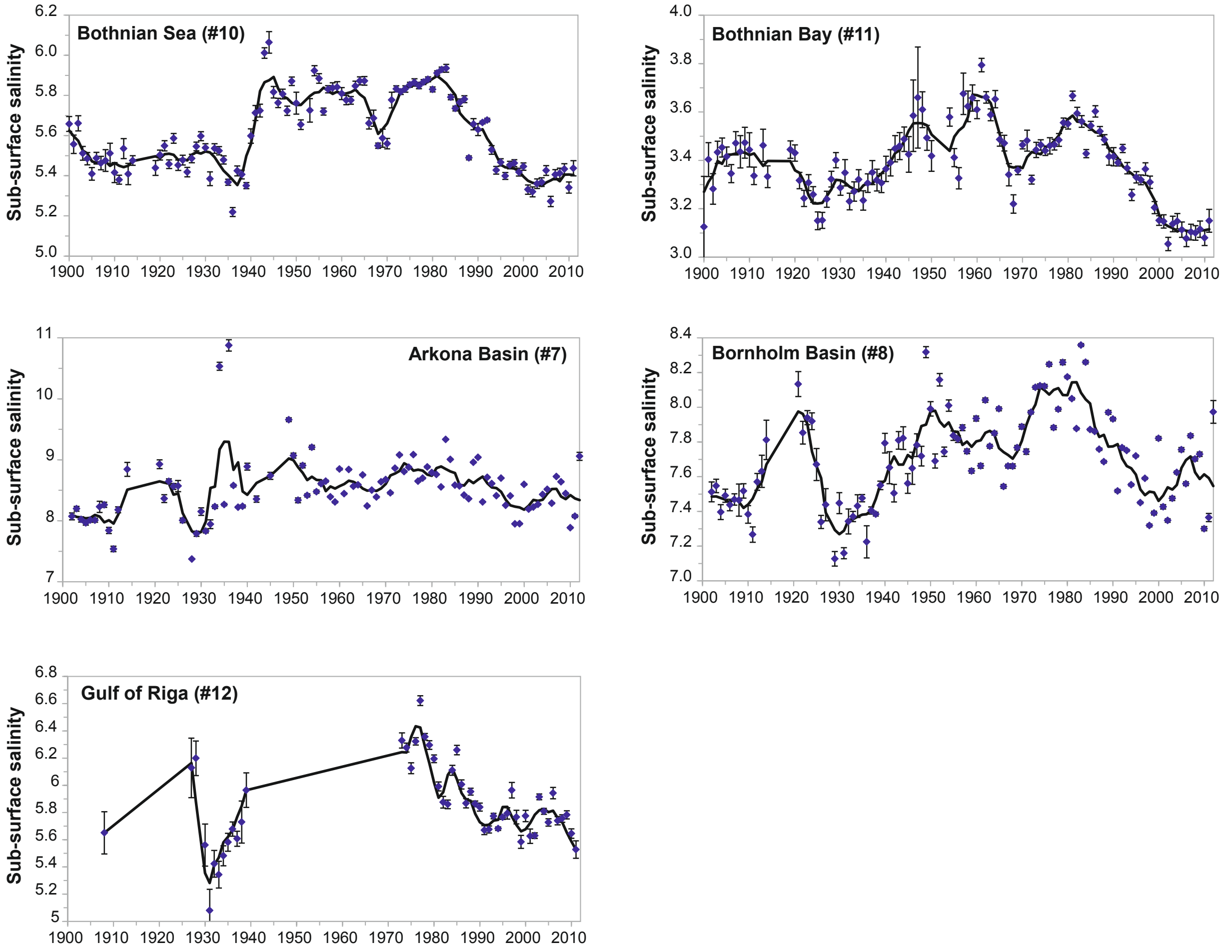


Fig. C.1: *Trends in sub-surface salinity (20-30 m, annual means) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Trends in salinity difference between sub-surface and bottom

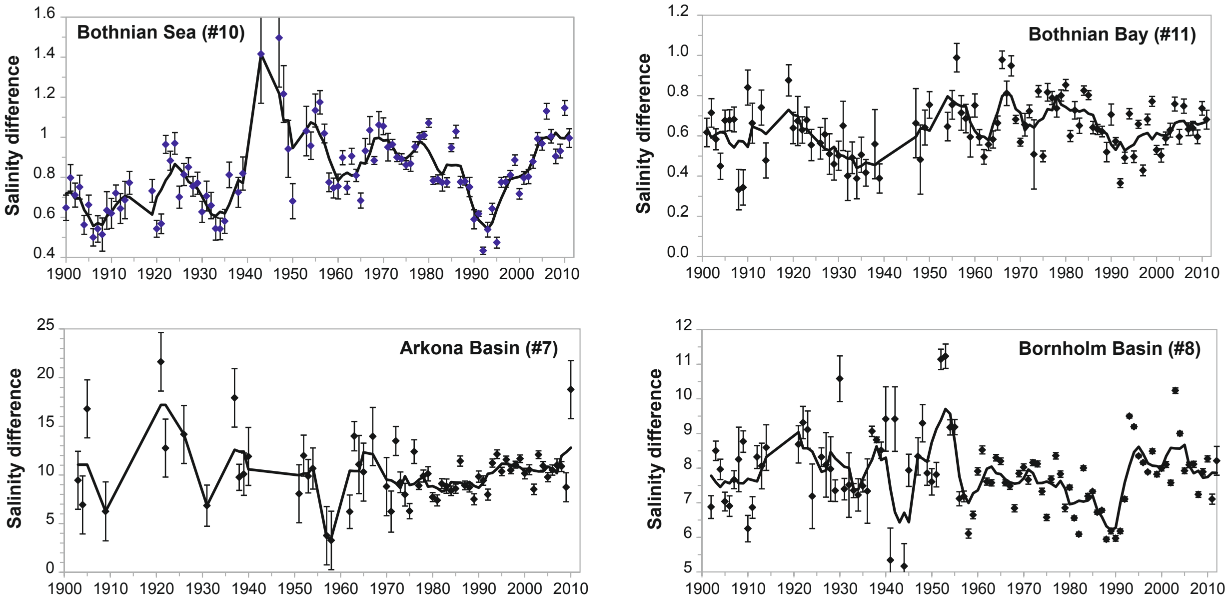


Fig. C.2: *Trends in salinity difference (annual means) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Trends in halocline depth

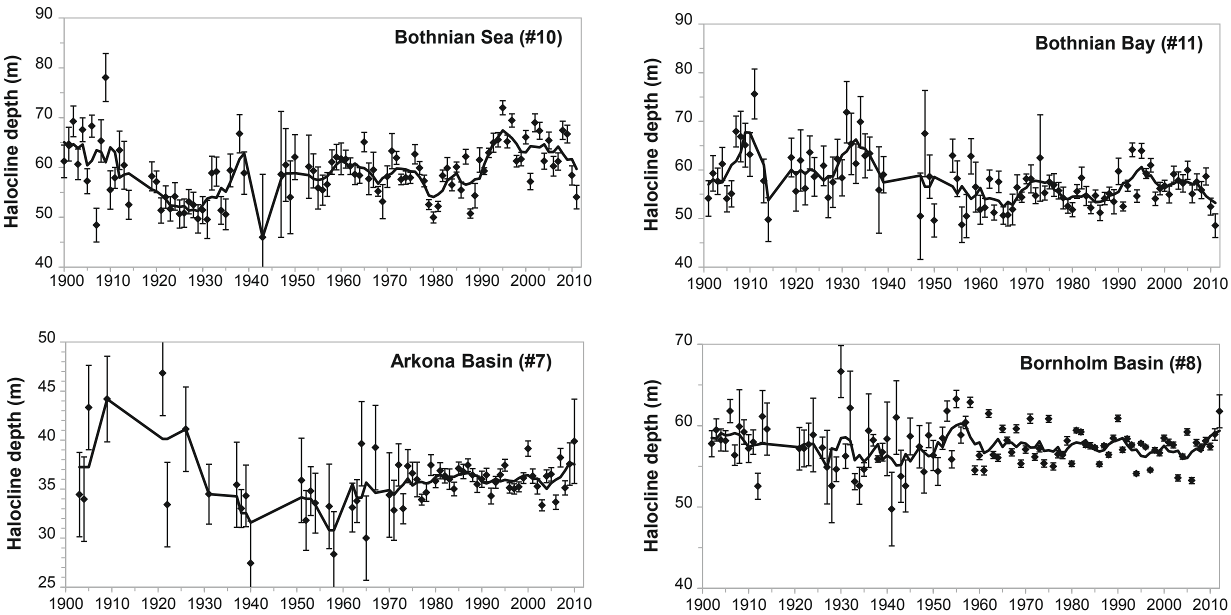


Fig. C.3: *Trends in halocline depth (annual means) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Trends in halocline steepness

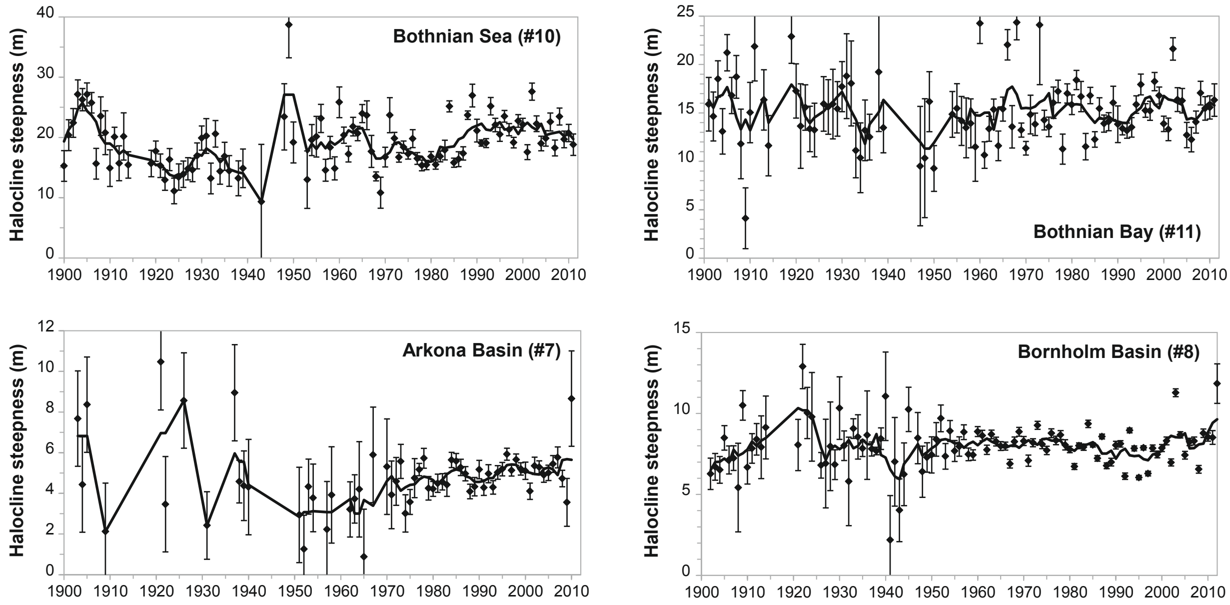


Fig. C.4: *Trends in halocline steepness (annual means) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Trends in integrated salinity

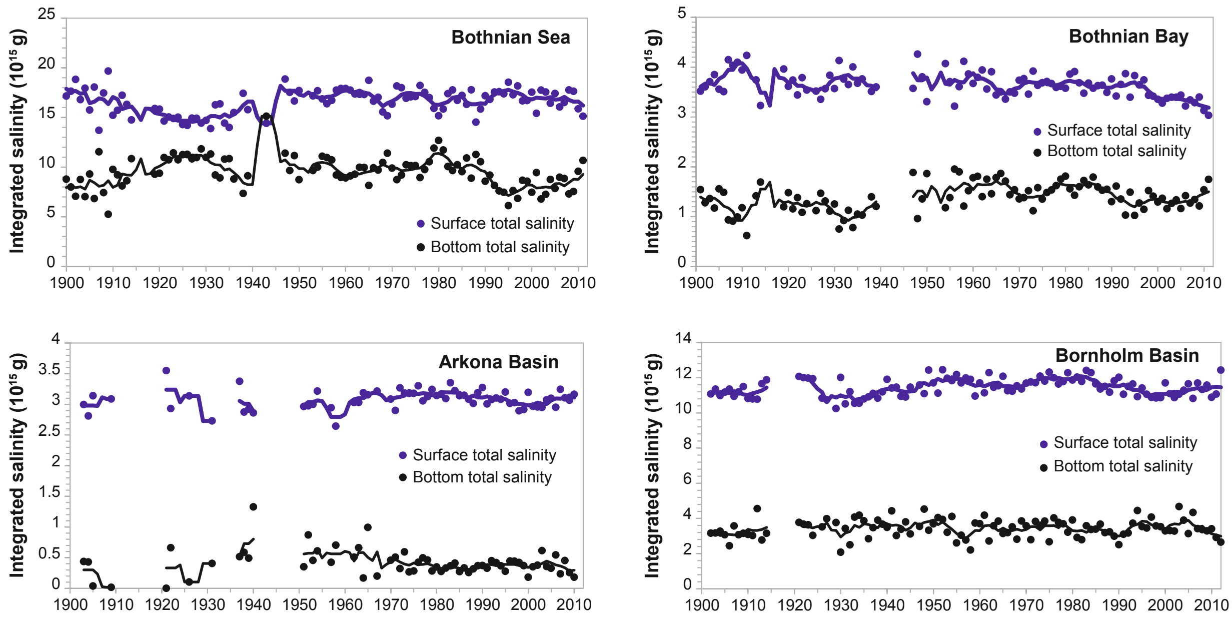


Fig. C.5: *Trends in total salinity content (annual means) with a five-year moving average (solid line).*

Seasonality in salinity parameters for the Arkona Basin

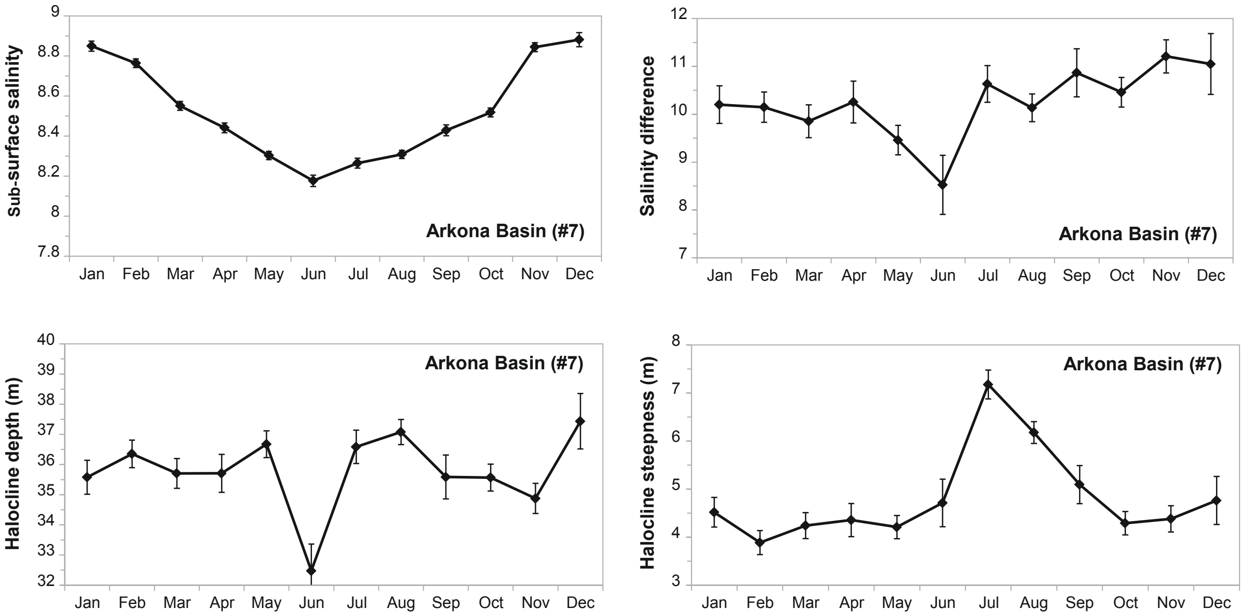


Fig. C.6: *Seasonal variation in salinity parameters for the Arkona Basin (1902-2010) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Seasonality in salinity parameters for the Bornholm Basin

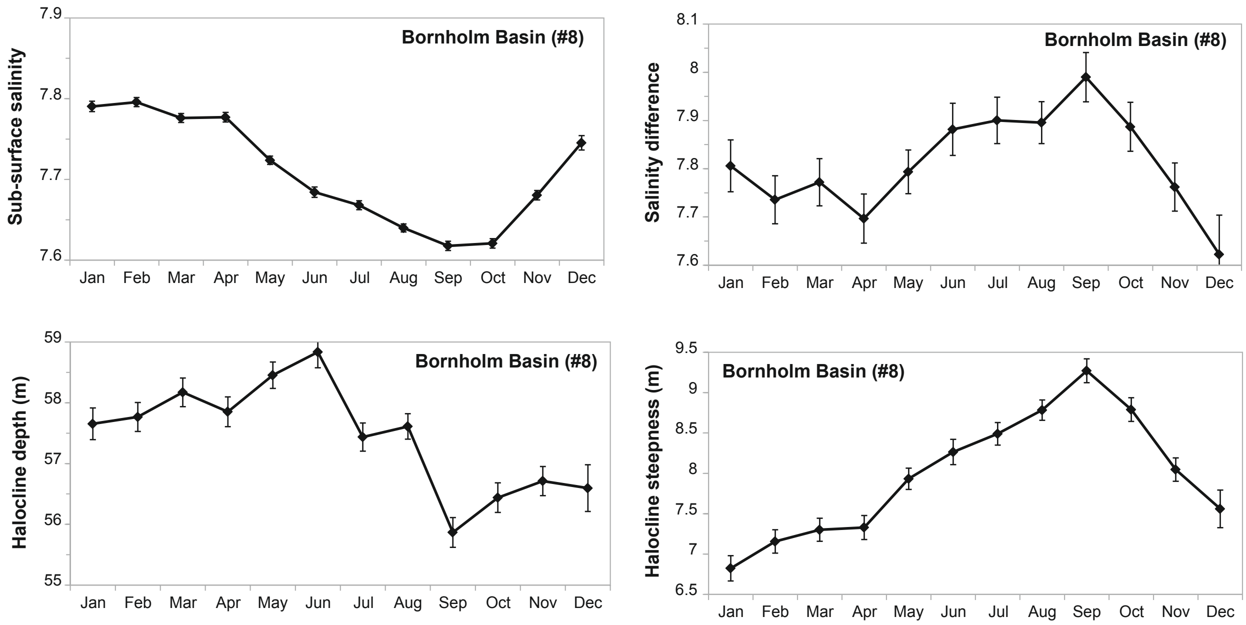


Fig. C.7: *Seasonal variation in salinity parameters for the Bornholm Basin (1902-2010) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Seasonality in salinity parameters for the Baltic Proper

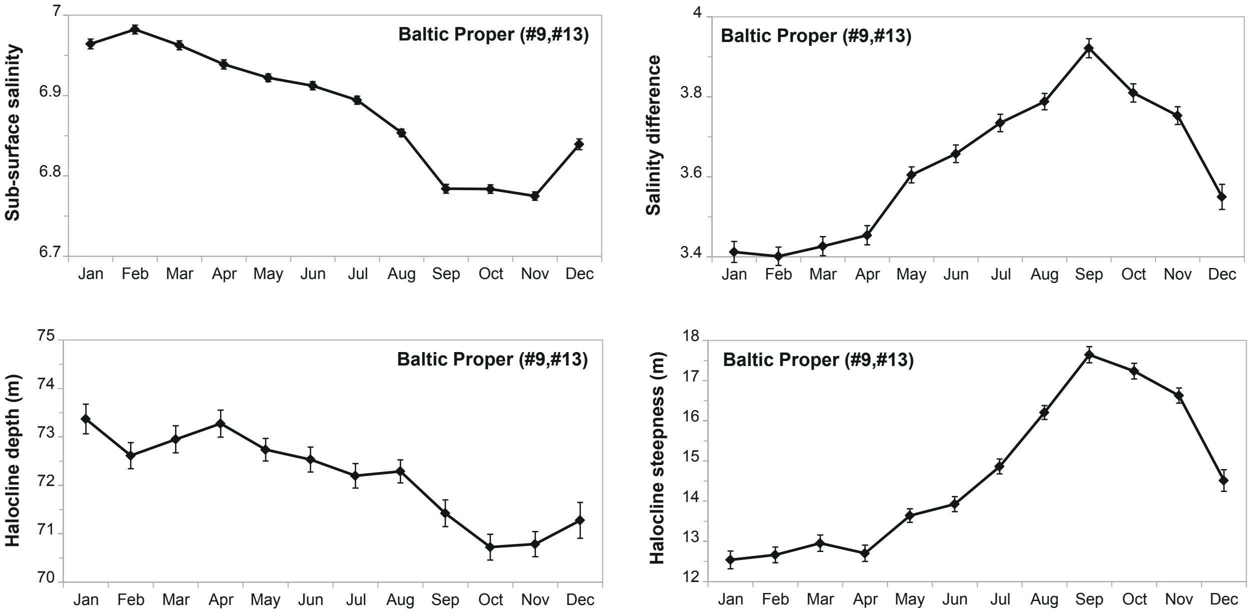


Fig. C.8: *Seasonal variation in salinity parameters for the Baltic Proper (1900-2010) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Seasonality in salinity parameters for the Bothnian Sea

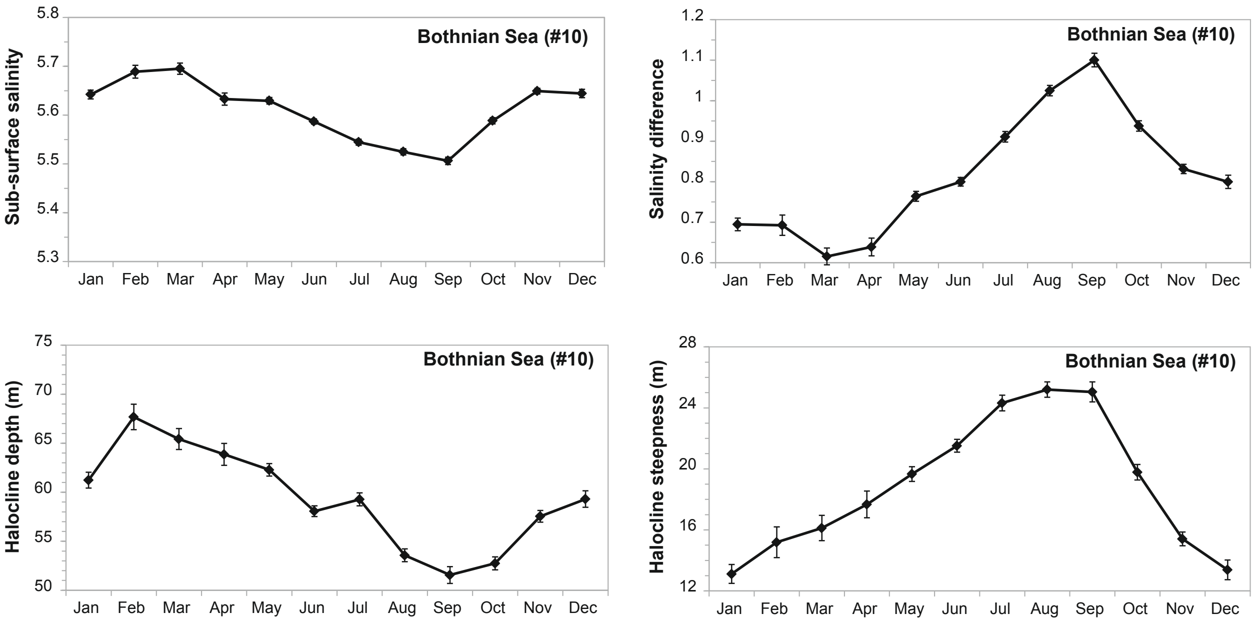


Fig. C.9: *Seasonal variation in salinity parameters for the Bothnian Sea (1900-2009) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Seasonality in salinity parameters for the Bothnian Bay

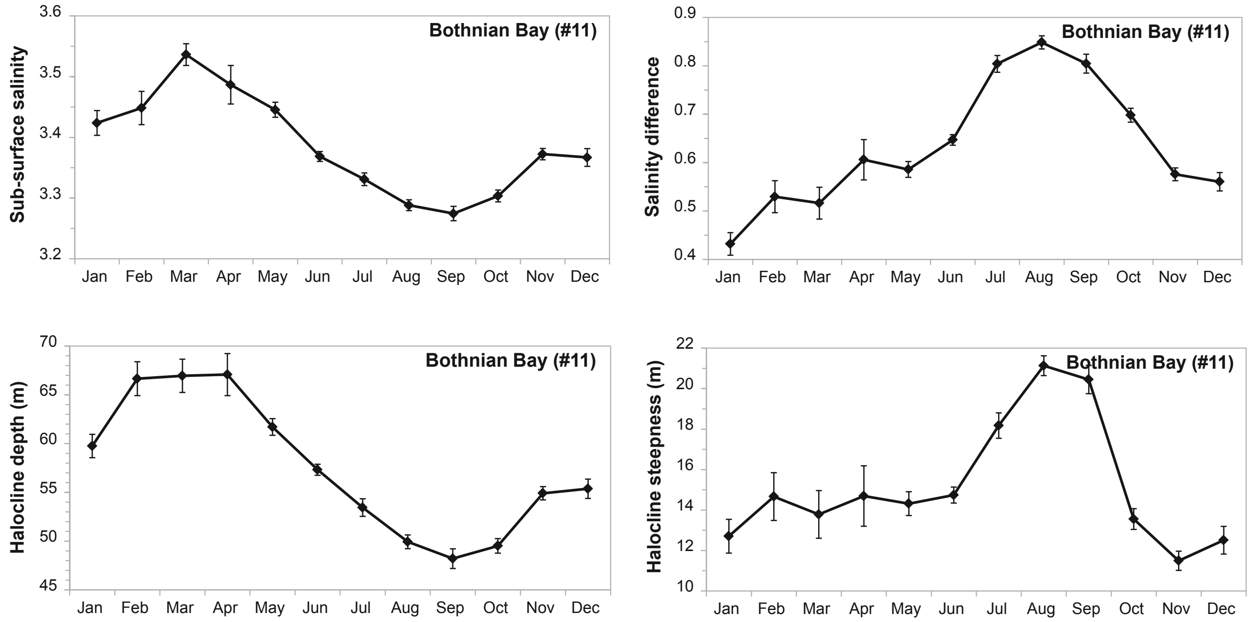


Fig. C.10: *Seasonal variation in salinity parameters for the Bothnian Sea (1900-2010) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Oxygen

Trends in oxygen debt below the halocline

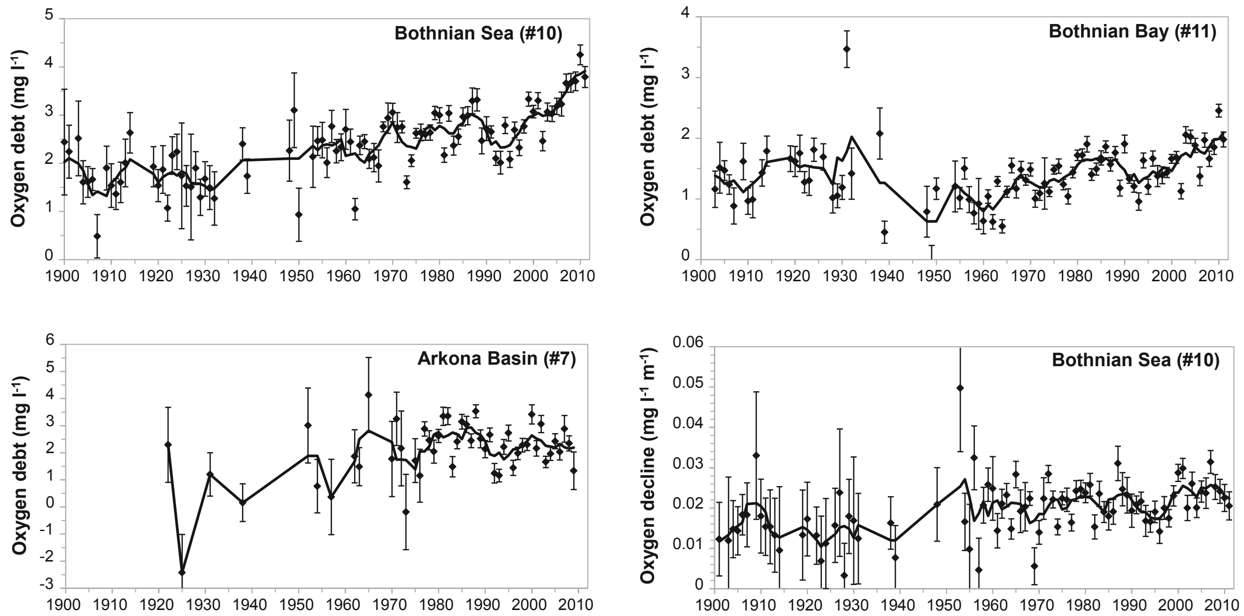
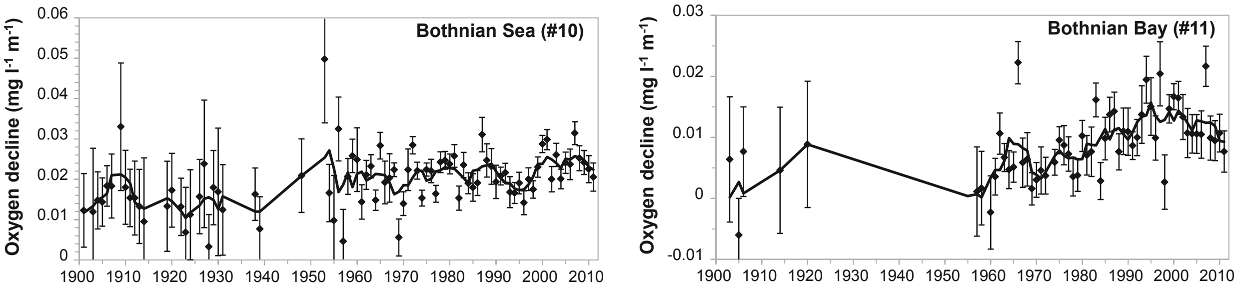


Fig. C.11: *Trends in oxygen debt below the halocline (annual means) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Trends in oxygen decline rate below the halocline



Not available for the Arkona Basin.

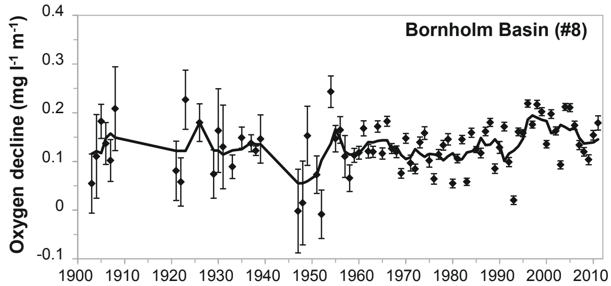
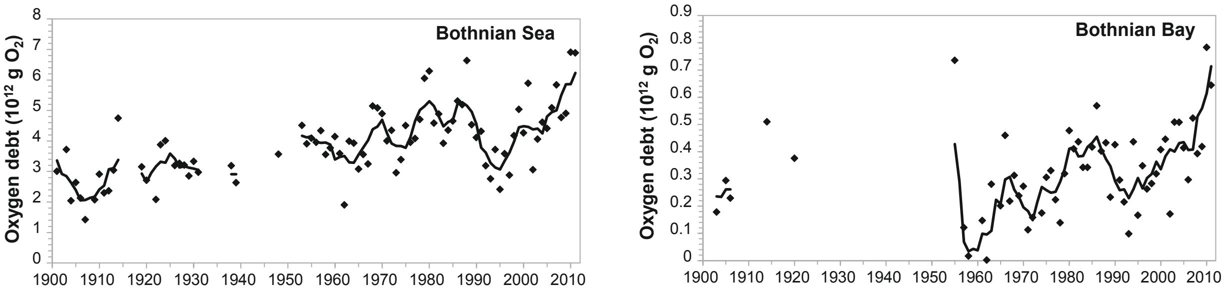


Fig. C.12: *Trends in oxygen decline rate below the halocline (annual means) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

Trends in total oxygen debt



Not available for the Arkona Basin.

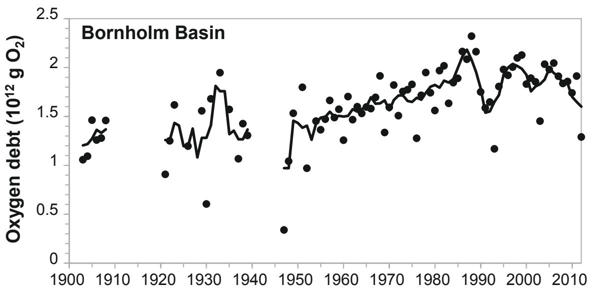
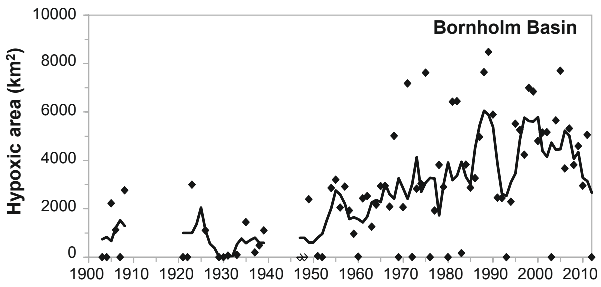


Fig. C.13: *Trends total oxygen debt below the halocline (annual means) with a five-year moving average (solid line).*

Trends in area and volume of hypoxia in Bornholm Basin

Hypoxia is shown for the Bornholm Basin only since hypoxia is not present in the Gulf of Bothnia and parameters are not available for the Arkona Basin (see above).



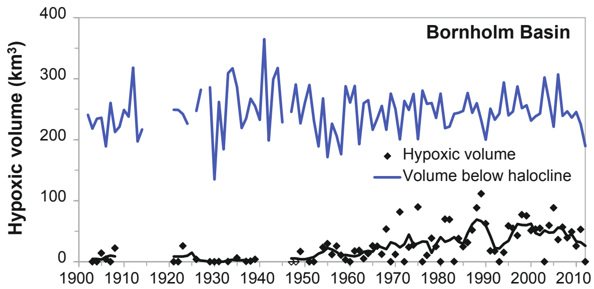
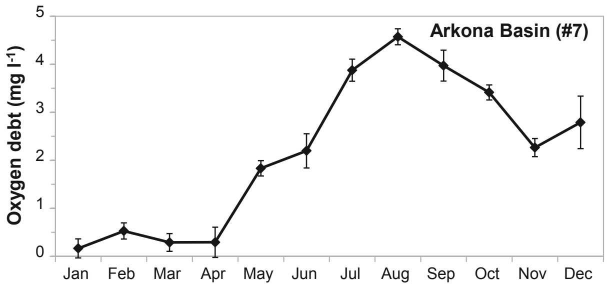


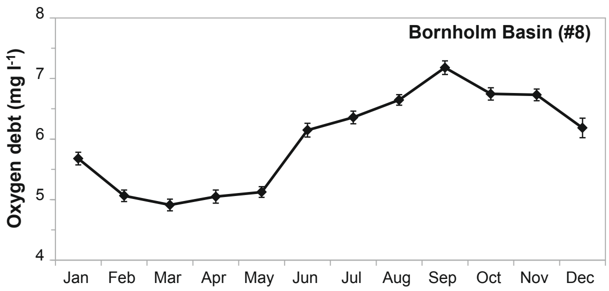
Fig. C.14: *Trends in area and volume of hypoxia in the Bornholm Basin (annual means) with a five-year moving average (solid line).*

Seasonality in oxygen parameters for the different basins



Not available for the Arkona Basin.

Fig. C.14 *Seasonal variation in oxygen parameters for the Arkona Basin (1902-2010) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*



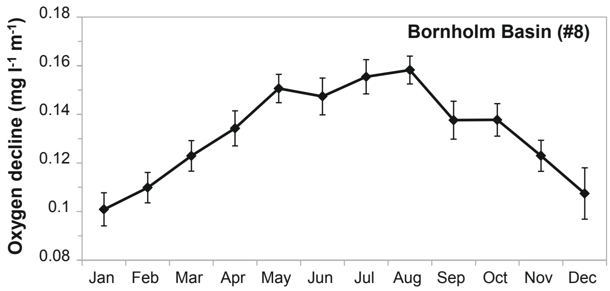


Fig. C.15 *Seasonal variation in oxygen parameters for the Bornholm Basin (1902-2010) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

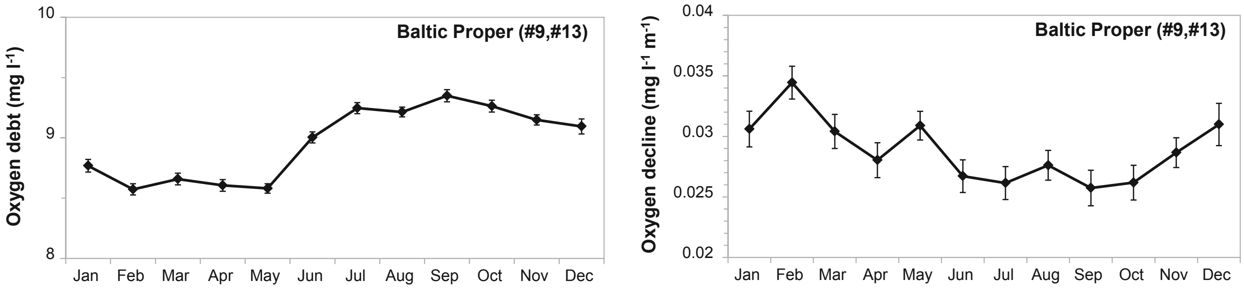


Fig. C.16 *Seasonal variation in oxygen parameters for the Bornholm Basin (1900-2010) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

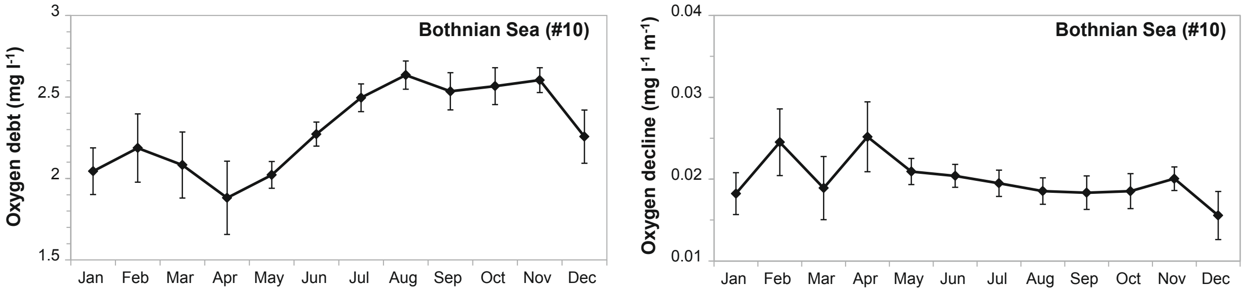


Fig. C.17 *Seasonal variation in oxygen parameters for the Bothnian Sea (1900-2009) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

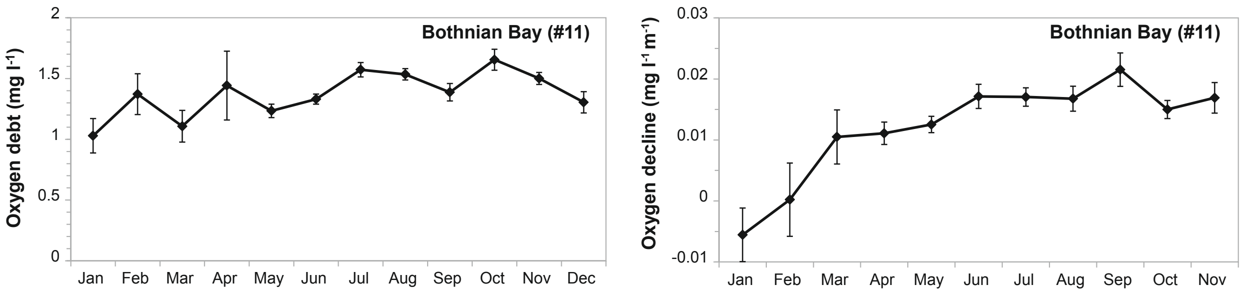


Fig. C.18 *Seasonal variation in oxygen parameters for the Bothnian Bay (1900-2009) with a five-year moving average (solid line). Error bars are 95% confidence intervals for the estimates.*

ANNEX D: Ensemble model results

Table D.1 *Estimated winter DIN levels (December-February, in µmol l-1) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	3.66	2.45	6.39	3.79	5.11	4.90	7.64	5.88
2. Central Kattegat	3.46	1.95	6.42	3.59	5.02	3.50	8.05	5.53
3. Southern Kattegat	3.27	1.59	4.73	2.82	4.73	2.79	6.86	4.79
4. Samsø Belt	3.06	1.40	6.45	3.25	4.86	2.73	8.56	5.38
5. Fehmarn Belt	2.27	2.16	4.22	2.58	3.91	6.65	6.82	5.79
6. The Sound	2.64	1.86	4.16	2.63	4.37	2.75	6.46	4.53
7. Arkona Basin	1.71	1.74	2.83	1.89	3.02	2.50	5.14	3.55
8. Bornholm Basin	1.84	1.95	3.16	2.11	3.42	2.73	6.68	4.28
9. Baltic Proper	2.16	2.65	2.50	2.22	4.31	3.31	3.93	3.85
10. Bothnian Sea	1.72	3.07	1.66	2.00	4.22	4.25	2.47	3.65
11. Bothnian Bay	2.77	6.27	1.52	3.46	6.29	5.73	2.56	4.86
12. Gulf of Riga	2.35	4.66	4.93	3.75	5.92	8.78	10.61	8.44
13. Gulf of Finland	3.77	5.08	5.32	4.37	6.51	8.88	10.52	8.64

Table D.2 *Estimated annual DIN levels (in µmol l-1) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	0.99	1.17	2.14	1.43	1.75	2.70	2.97	2.47
2. Central Kattegat	0.91	0.86	2.00	1.26	1.65	1.62	2.94	2.07
3. Southern Kattegat	0.87	0.62	1.37	0.95	1.55	1.17	2.54	1.75
4. Samsø Belt	0.77	0.61	1.99	1.13	1.53	1.12	3.18	1.94
5. Fehmarn Belt	0.60	0.83	1.24	0.89	1.32	3.09	2.71	2.37
6. The Sound	0.83	0.83	1.48	1.04	1.69	1.25	2.70	1.88
7. Arkona Basin	0.62	0.67	0.87	0.72	1.25	1.05	2.02	1.44
8. Bornholm Basin	0.74	0.79	1.26	0.93	1.50	1.28	3.51	2.09
9. Baltic Proper	1.00	1.55	1.04	1.20	2.21	2.00	2.00	2.07
10. Bothnian Sea	1.05	1.64	0.79	1.16	2.58	2.34	1.37	2.10
11. Bothnian Bay	2.58	6.53	0.88	3.33	5.94	5.08	1.63	4.22
12. Gulf of Riga	1.22	5.12	2.14	2.83	3.41	9.53	6.03	6.32
13. Gulf of Finland	2.15	3.98	2.94	3.02	4.16	7.18	6.78	6.04

Table D.3 *Estimated winter DIP levels (December-February, in µmol l^{-1}) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	0.41	0.15	0.64	0.40	0.58	0.19	0.74	0.50
2. Central Kattegat	0.41	0.19	0.66	0.42	0.61	0.24	0.80	0.55
3. Southern Kattegat	0.42	0.20	0.56	0.39	0.64	0.28	0.73	0.55
4. Samsø Belt	0.43	0.28	0.64	0.45	0.70	0.42	0.86	0.66
5. Fehmarn Belt	0.33	0.40	0.60	0.44	0.60	0.60	0.86	0.69
6. The Sound	0.33	0.25	0.45	0.35	0.59	0.42	0.67	0.56
7. Arkona Basin	0.21	0.28	0.36	0.28	0.50	0.51	0.62	0.54
8. Bornholm Basin	0.21	0.27	0.28	0.26	0.55	0.54	0.57	0.55
9. Baltic Proper	0.28	0.24	0.22	0.25	0.80	0.47	0.45	0.57
10. Bothnian Sea	0.19	0.23	0.21	0.21	0.49	0.61	0.34	0.48
11. Bothnian Bay	0.06	0.09	0.10	0.08	0.12	0.39	0.19	0.23
12. Gulf of Riga	0.30	0.12	0.23	0.22	1.08	0.29	0.70	0.69
13. Gulf of Finland	0.45	0.28	0.33	0.35	1.09	0.61	0.81	0.84

Table D.4 *Estimated annual DIP levels (in µmol l^{-1}) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	0.20	0.15	0.28	0.21	0.26	0.19	0.32	0.26
2. Central Kattegat	0.21	0.19	0.29	0.23	0.29	0.24	0.36	0.29
3. Southern Kattegat	0.22	0.20	0.23	0.22	0.32	0.28	0.30	0.30
4. Samsø Belt	0.23	0.28	0.29	0.27	0.36	0.42	0.39	0.39
5. Fehmarn Belt	0.18	0.40	0.26	0.28	0.32	0.60	0.38	0.43
6. The Sound	0.18	0.25	0.21	0.22	0.33	0.42	0.33	0.36
7. Arkona Basin	0.13	0.28	0.15	0.19	0.33	0.51	0.27	0.37
8. Bornholm Basin	0.13	0.27	0.12	0.17	0.36	0.54	0.27	0.39
9. Baltic Proper	0.17	0.24	0.10	0.17	0.51	0.47	0.22	0.40
10. Bothnian Sea	0.13	0.23	0.11	0.15	0.33	0.61	0.19	0.37
11. Bothnian Bay	0.04	0.09	0.06	0.07	0.08	0.39	0.11	0.20
12. Gulf of Riga	0.20	0.12	0.10	0.14	0.73	0.29	0.35	0.46
13. Gulf of Finland	0.30	0.28	0.18	0.25	0.77	0.61	0.47	0.62

Table D.5 *Estimated summer Chl a levels (Jun-Sep, in µg l^{-1}) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	0.38	1.35	0.46	0.73	0.67	2.37	0.69	1.25
2. Central Kattegat	0.50	1.03	0.43	0.65	1.02	2.52	0.67	1.40
3. Southern Kattegat	0.54	0.88	0.40	0.61	1.25	2.37	0.66	1.43
4. Samsø Belt	0.84	1.18	0.66	0.89	2.18	2.82	0.95	1.98
5. Fehmarn Belt	0.53	1.92	0.52	0.99	1.56	3.31	0.85	1.91
6. The Sound	0.57	2.23	1.01	1.27	1.57	3.88	1.33	2.26
7. Arkona Basin	0.30	2.34	0.53	1.06	1.53	3.92	0.97	2.14
8. Bornholm Basin	0.29	2.63	0.52	1.15	1.53	4.31	1.14	2.32
9. Baltic Proper	0.25	1.47	0.37	0.69	1.25	3.05	0.87	1.72
10. Bothnian Sea	0.14	1.07	0.59	0.60	0.40	3.42	1.03	1.62
11. Bothnian Bay	0.04	0.64	0.55	0.41	0.12	2.14	1.28	1.18
12. Gulf of Riga	0.32	0.62	0.65	0.53	1.66	1.13	1.48	1.42
13. Gulf of Finland	0.59	1.47	1.39	1.15	1.91	2.63	3.80	2.78

Table D.6 *Estimated annual Chl a levels (in µg l^{-1}) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	0.31	1.61	1.30	1.07	0.63	2.46	1.72	1.61
2. Central Kattegat	0.33	1.66	1.44	1.14	0.75	2.86	1.91	1.84
3. Southern Kattegat	0.35	1.27	1.19	0.94	0.84	2.47	1.66	1.66
4. Samsø Belt	0.48	1.63	1.58	1.23	1.22	3.15	2.09	2.15
5. Fehmarn Belt	0.33	2.21	1.29	1.28	0.96	3.74	1.93	2.21
6. The Sound	0.38	1.86	1.36	1.20	1.05	3.08	1.79	1.97
7. Arkona Basin	0.19	1.75	1.01	0.98	0.83	2.86	1.61	1.77
8. Bornholm Basin	0.18	1.85	1.01	1.01	0.84	2.93	1.70	1.82
9. Baltic Proper	0.16	1.17	0.79	0.71	0.61	2.11	1.30	1.34
10. Bothnian Sea	0.07	0.95	0.71	0.58	0.22	2.15	1.01	1.13
11. Bothnian Bay	0.01	0.40	0.47	0.30	0.04	1.32	0.91	0.76
12. Gulf of Riga	0.19	0.90	1.31	0.80	0.93	1.84	2.77	1.85
13. Gulf of Finland	0.29	1.39	1.59	1.09	0.95	2.35	3.27	2.19

Table D.7 *Estimated summer Secchi depth levels (Jun-Sep, in m) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	9.13	10.50	10.46	10.03	8.31	9.67	9.29	9.09
2. Central Kattegat	8.79	10.19	10.24	9.74	7.76	9.25	8.86	8.62
3. Southern Kattegat	8.53	11.40	10.33	10.09	7.39	10.09	8.82	8.76
4. Samsø Belt	7.84	10.58	9.66	9.36	6.38	9.43	8.10	7.97
5. Fehmarn Belt	7.88	10.42	10.05	9.45	6.73	9.32	8.29	8.11
6. The Sound	7.57	10.17	9.48	9.07	6.38	8.86	7.84	7.69
7. Arkona Basin	6.85	10.16	10.09	9.03	5.79	8.96	8.16	7.64
8. Bornholm Basin	6.72	9.99	9.95	8.89	5.67	8.79	7.99	7.48
9. Baltic Proper	6.72	10.92	9.95	9.20	5.87	9.66	8.01	7.85
10. Bothnian Sea	5.88	11.17	10.11	9.05	5.63	9.21	8.07	7.64
11. Bothnian Bay	4.78	11.19	9.66	8.54	4.71	9.90	7.32	7.31
12. Gulf of Riga	5.63	11.10	8.59	8.44	4.90	10.81	6.56	7.42
13. Gulf of Finland	5.42	10.81	6.65	7.62	4.61	9.93	5.32	6.62

Table D.8 *Estimated annual Secchi depth levels (m) from a 2D model (BALTSEM) and two 3D models (ERGOM and MIKE) of the pre-eutrophication period (1900) and the baseline (1997-2006).*

Basin	Pre-eutrophication 1900				Baseline 1997-2006			
	BALTSEM	ERGOM	MIKE	Average	BALTSEM	ERGOM	MIKE	Average
1. Northern Kattegat	9.46	10.28	9.91	9.88	8.67	9.57	8.96	9.07
2. Central Kattegat	9.35	9.83	9.62	9.60	8.48	8.99	8.53	8.67
3. Southern Kattegat	9.19	11.06	9.75	10.00	8.30	9.99	8.46	8.91
4. Samsø Belt	8.82	10.13	9.10	9.35	7.78	9.02	7.84	8.21
5. Fehmarn Belt	8.56	10.06	9.46	9.36	7.68	8.92	7.95	8.18
6. The Sound	8.22	10.20	9.31	9.25	7.32	9.20	7.87	8.13
7. Arkona Basin	7.39	10.48	9.62	9.16	6.70	9.54	7.93	8.06
8. Bornholm Basin	7.14	10.39	9.48	9.00	6.45	9.43	7.77	7.89
9. Baltic Proper	7.05	10.96	9.51	9.18	6.57	10.12	7.88	8.19
10. Bothnian Sea	6.13	10.94	9.74	8.93	5.97	9.94	8.07	7.99
11. Bothnian Bay	4.88	11.34	9.34	8.52	4.85	10.41	7.41	7.56
12. Gulf of Riga	6.08	10.70	8.02	8.26	5.58	9.81	6.41	7.27
13. Gulf of Finland	6.02	10.72	6.60	7.78	5.57	9.96	5.55	7.03

ANNEX E: Distribution of nutrients and Chlorophyll *a* from the earliest period and reference period

Table E.1: *Average of annual total nitrogen (TN) and winter (December-February) TN during the earliest data period (1970 – 1975) and baseline period (1997 – 2006). Standard deviation and percentiles describe the interannual variation in annual and seasonal means within the period given.*

TN (µmol l^{-1})	Yearly means				Winter means			
	Earliest data		baseline		Earliest data		baseline	
Basin	**Avg**	**Std**	**Avg**	**Std**	**Avg**	**Std**	**Avg**	**Std**
Arkona Basin	17.36	2.38	20.95	1.02	15.82	4.12	22.10	1.39
Baltic Proper	16.23	0.66	20.78	0.88	15.79	4.29	20.98	0.53
Bornholm Basin	16.29	0.98	21.50	0.93	17.38	3.02	22.26	1.39
Bothnian Bay	16.88	2.02	19.16	0.59	18.88	0.83	19.51	0.79
Bothnian Sea	15.66	1.48	17.13	0.32	17.67	1.40	18.18	0.59
Danish Straits	21.79	3.05	20.59	1.03	26.04	3.49	22.44	1.00
Gulf of Finland	22.15	1.82	23.75	2.60	33.12	4.67	30.87	3.64
Gulf of Riga[1]	37.98	6.42	29.31	2.64	37.06	3.94	32.04	3.31
Kattegat	17.43	2.37	18.50	1.61	16.94	4.42	20.99	1.74

1 Earliest data covers 1990-1995 due to the lack of older data.

Table E.2: *Average of annual dissolved inorganic nitrogen (DIN) and winter (December-February) DIN during earliest data period (1970 – 1975) and baseline period (1997 – 2006). Standard deviation and percentiles describe the interannual variation in annual and seasonal means within the period given.*

DIN (µmol l^{-1})	Yearly means				Winter means			
	Earliest data		baseline		Earliest data		baseline	
Basin	**Avg**	**Std**	**Avg**	**Std**	**Avg**	**Std**	**Avg**	**Std**
Arkona Basin	1.34	0.38	1.27	0.39	3.34	1.02	5.03	1.33
Baltic Proper	1.70	0.40	0.89	0.16	3.01	0.51	3.68	0.45
Bornholm Basin	1.35	0.39	1.15	0.32	3.29	1.11	3.70	0.74
Bothnian Bay	5.05	2.29	5.97	0.46	6.71	2.02	7.35	0.40
Bothnian Sea	1.98	0.67	0.96	0.17	3.51	0.63	3.60	0.53
Danish Straits	2.38	1.03	1.63	0.44	7.40	1.38	7.27	1.43
Gulf of Finland	3.68	0.96	2.25	0.65	9.22	1.37	10.37	2.21
Gulf of Riga[1]	5.59	2.37	2.29	1.11	9.86	1.25	9.00	2.03
Kattegat	1.83	0.44	1.16	0.18	4.75	1.72	6.27	1.42

[1]Earliest data covers 1990-1995 due to the lack of older data.

Table E.3: *Average of annual total phosphorus (TP) and winter (December-February) TP during earliest data period (1970 – 1975) and baseline period (1997 – 2006). Standard deviation and percentiles describe the interannual variation in annual and seasonal means within the period given.*

TP (μmol l^{-1})	Yearly means				Winter means			
	Earliest data		baseline		Earliest data		baseline	
Basin	Avg	Std	Avg	Std	Avg	Std	Avg	Std
Arkona Basin	0.66	0.05	0.70	0.11	0.75	0.18	0.84	0.11
Baltic Proper	0.44	0.04	0.59	0.06	0.60	0.08	0.72	0.12
Bornholm Basin	0.57	0.07	0.70	0.09	0.66	0.10	0.82	0.16
Bothnian Bay	0.18	0.05	0.17	0.01	0.20	0.03	0.17	0.02
Bothnian Sea	0.24	0.03	0.32	0.03	0.33	0.04	0.39	0.06
Danish Straits	0.97	0.10	0.71	0.07	1.12	0.17	0.95	0.09
Gulf of Finland	0.56	0.10	0.87	0.11	1.16	0.15	1.26	0.21
Gulf of Riga[1]	0.71	0.09	0.84	0.10	0.90	0.13	1.15	0.13
Kattegat	0.64	0.11	0.57	0.05	0.90	0.08	0.80	0.07

[1]Earliest data covers 1973-1980 due to the lack of older data.

Table E.4: *Average of annual dissolved inorganic phosphorus (DIP) and winter (December-February) DIP during earliest data period (1970 – 1975) and baseline period (1997 – 2006). Standard deviation describes the interannual variation in annual and seasonal means within the period given.*

DIP (μmol l^{-1})	Yearly means				Winter means			
	Earliest data		baseline		Earliest data		baseline	
Basin	Avg	Std	Avg	Std	Avg	Std	Avg	Std
Arkona Basin	0.26	0.06	0.27	0.11	0.47	0.15	0.52	0.10
Baltic Proper	0.18	0.03	0.21	0.04	0.33	0.08	0.49	0.10
Bornholm Basin	0.20	0.07	0.32	0.08	0.36	0.08	0.56	0.12
Bothnian Bay	0.07	0.02	0.05	0.01	0.06	0.00	0.05	0.01
Bothnian Sea	0.11	0.02	0.09	0.01	0.18	0.02	0.20	0.03
Danish Straits	0.39	0.06	0.23	0.05	0.74	0.17	0.60	0.07
Gulf of Finland	0.29	0.07	0.36	0.11	0.95	0.09	1.01	0.23
Gulf of Riga[1]	0.37	0.13	0.20	0.05	0.60	0.13	0.78	0.15
Kattegat	0.20	0.06	0.15	0.02	0.58	0.10	0.49	0.07

[1]Earliest data covers 1973-1980 due to the lack of older data.

Table E.5: *Average of annual Chlorophyll a (Chl a) and summer (June-September) Chl a during the earliest data period (1972 – 1980) and baseline period (1997 – 2006). Standard deviation describes the interannual variation in annual and seasonal means within the period given.*

Chl *a* (µg l^{-1})	Yearly means				Summer means			
	Earliest data		baseline		Earliest data		baseline	
Basin	Avg	Std	Avg	Std	Avg	Std	Avg	Std
Arkona Basin	1.36	0.25	1.70	0.14	1.44	0.28	1.93	0.32
Baltic Proper	0.93	0.18	1.84	0.18	1.74	0.28	2.79	0.39
Bornholm Basin	1.20	0.26	1.97	0.23	2.44	2.06	2.24	0.44
Bothnian Bay[1]	1.23	0.20	1.60	0.45	1.63	0.30	2.26	0.65
Bothnian Sea[1]	1.33	0.72	2.08	0.61	1.52	0.79	2.38	0.69
Danish Straits	1.79	0.19	2.17	0.22	1.89	0.34	2.32	0.30
Gulf of Finland	2.54	0.79	3.31	0.45	4.37	2.60	4.98	0.64
Gulf of Riga[2]	5.10	0.31	4.47	1.64	4.12	0.30	4.26	1.68
Kattegat	1.45	0.28	1.75	0.21	1.22	0.29	1.35	0.22

[1]Earliest data covers 1979-1985 due to lack of older data.
[2]Earliest data covers 1990-1995 due to lack of older data

Table E.6: *Average of annual Secchi depth and summer (June-September) Secchi depth from before 1940 (earliest data) and the period 1997 – 2006 (baseline). Standard deviation describes the interannual variation in annual and seasonal means within the period given.*

Secchi depth (m)	Yearly means				Summer means			
	Earliest data		baseline		Earliest data		baseline	
Basin	Avg	Std	Avg	Std	Avg	Std	Avg	Std
Arkona Basin	9.21	1.13	8.20	0.72	9.14	1.19	7.33	0.79
Baltic Proper	11.06	1.61	7.43	0.26	9.75	1.50	5.72	0.34
Bornholm Basin	9.98	1.04	7.93	0.41	8.84	1.15	6.59	0.43
Bothnian Bay	8.48	1.43	4.70	1.32	7.92	1.32	4.48	0.66
Bothnian Sea	9.29	1.64	4.95	0.60	9.19	1.72	4.93	0.61
Danish Straits	7.74	1.09	7.19	0.48	8.79	1.15	6.78	0.69
Gulf of Finland	6.84	0.99	3.73	0.75	7.04	1.07	3.27	0.27
Gulf of Riga	4.84	0.32	3.30	0.41	4.29	0.41	3.28	0.45
Kattegat	9.85	2.02	6.60	0.39	9.59	0.89	7.48	0.48

ANNEX F: Rescaling targets for Secchi depth, nutrients and Chlorophyll *a* to HELCOM's sub-divisions

The proposed targets for the BALTSEM basins have been recalculated into HELCOM's spatial sub-division by scaling the basin-specific targets (Tables 4.1 and 4.2) with the spatial models (Fig. 2.2, 2.5, 2.8, 2.11, 2.14, and 2.17) to produce spatially distributed targets for Secchi depth (Fig. F.1); TN and TP (Fig. F.2); DIN (Fig. F.3); DIP (Fig. F.4); and Chl *a* (Fig. F.5). These spatial distributions were subsequently used to calculate targets for HELCOM's spatial sub-divisions by averaging the spatial distributions for the different sub-divisions.

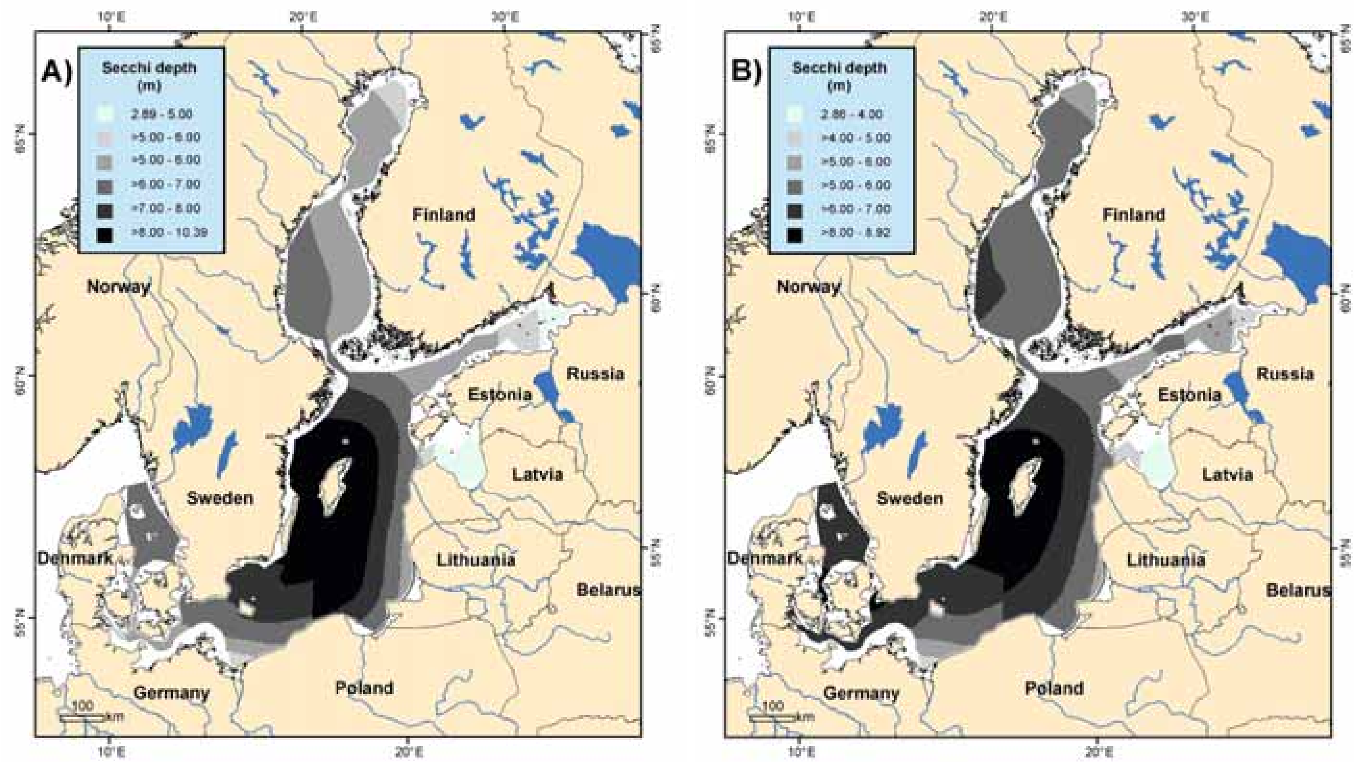


Figure F.1 *Spatial distribution of targets for Secchi depth annual means (A) and summer means (B, Jun-Sep). Targets were found from basin-specific time series, which may cause discontinuities across basin boundaries.*

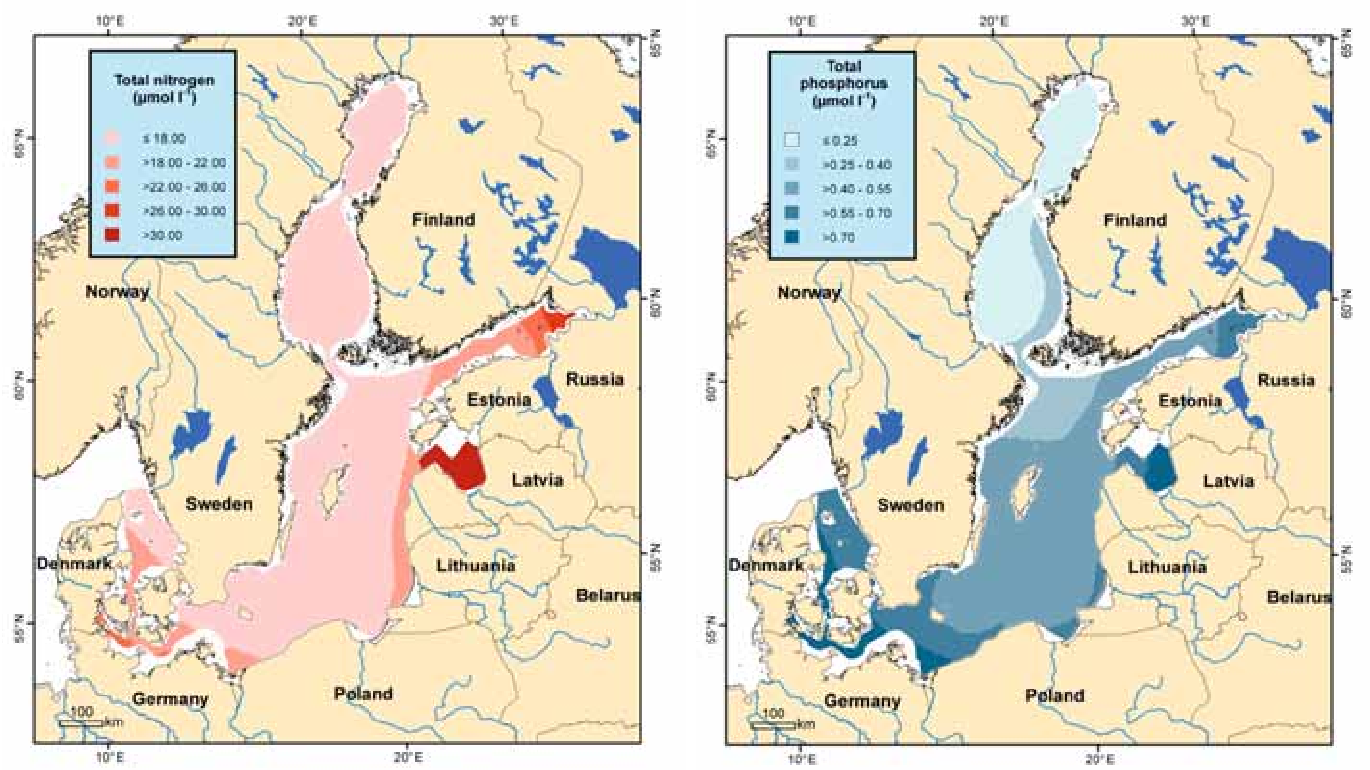


Figure F.2 *Spatial distribution of targets for annual TN and TP (in µmol l-1). Targets were found from basin-specific time series, which may cause discontinuities across basin boundaries.*

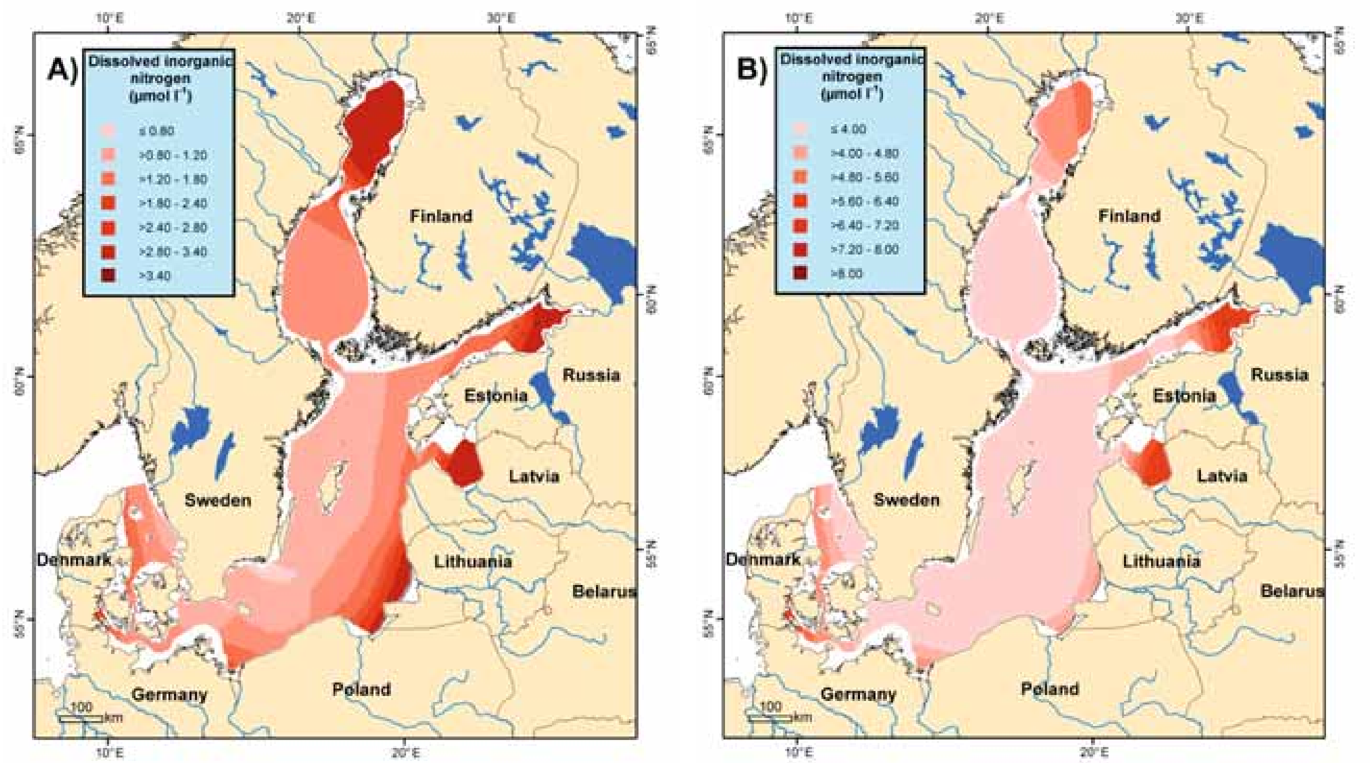


Figure F.3 *Spatial distribution of targets for DIN (in µmol l-1) annual means (A) and winter means (B, Dec-Feb). Targets were found from basin-specific time series, which may cause discontinuities across basin boundaries.*

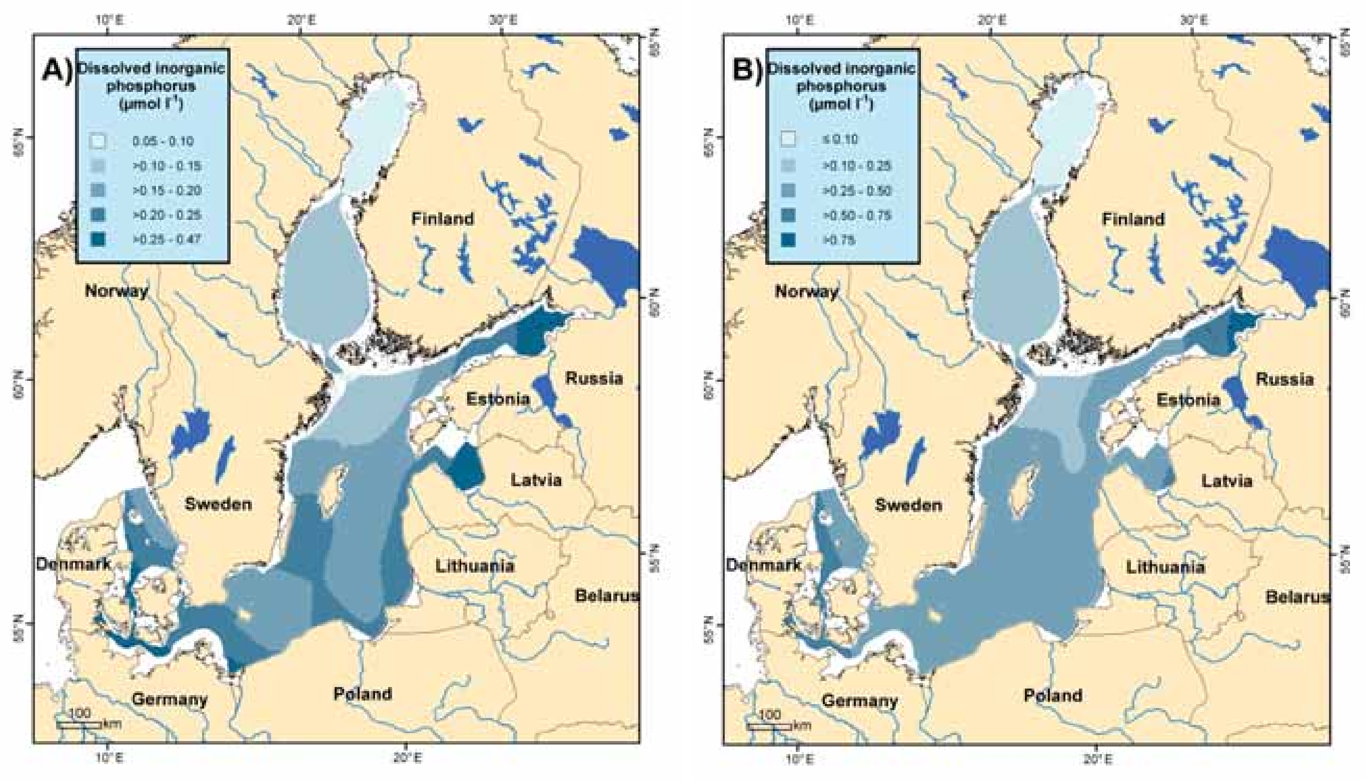


Figure F.4 *Spatial distribution of targets for DIP (in µmol l-1) annual means (A) and winter means (B, Dec-Feb). Targets were found from basin-specific time series, which may cause discontinuities across basin boundaries.*

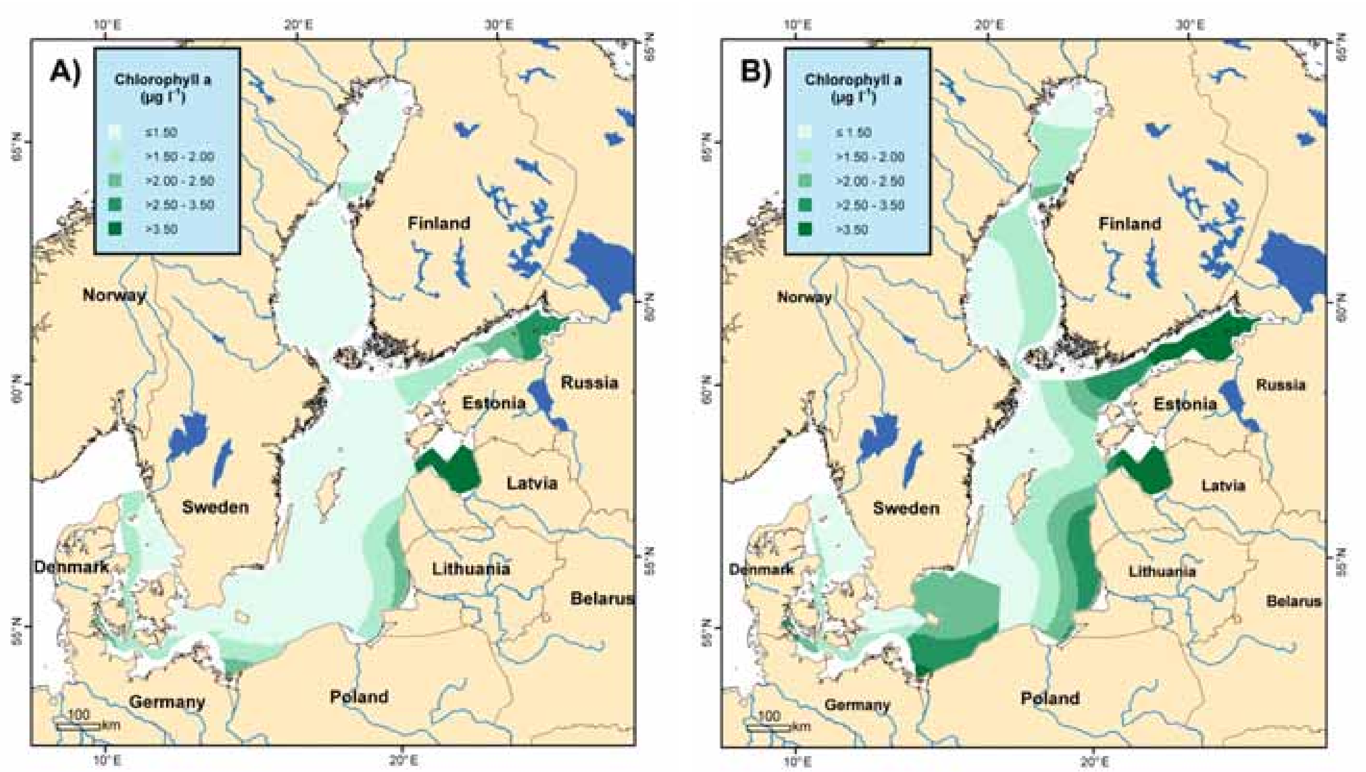


Figure F.5 *Spatial distribution of targets for Chl a (in µg l-1) annual means (A) and summer means (B, Jun-Sep). Targets were found from basin-specific time series, which may cause discontinuities across basin boundaries.*

Table F.1 *Secchi depth targets (Table 4.1) recalculated from the spatial model (Fig. F.1) to the HELCOM sub-divisions. Summer targets are Jun-Sep.*

Basin	Secchi depth target (m)	
	Summer	Annual
Kattegat	7.56	7.52
The Sound	8.17	7.52
Great Belt	8.46	6.62
Little Belt	7.31	5.76
Kiel Bay	7.41	6.11
Bay of Mecklenburg	7.43	6.42
Gdansk Basin	6.52	7.82
Arkona Sea	7.17	7.26
Bornholm Sea	7.11	8.43
Eastern Gotland Basin	7.55	8.82
Western Gotland Basin	8.43	9.98
Northern Baltic Proper	7.12	8.05
Gulf of Riga	3.82	3.98
Gulf of Finland	5.50	5.55
Åland Sea	6.85	7.34
Bothnian Sea	6.79	6.89
The Quark	6.51	6.49
Bothnian Bay	6.30	6.38

Table F.2 *Targets for nutrients (in µmol l^{-1}) and Chlorophyll a (in µg l^{-1}) recalculated from the targets in Table 4.2 and the spatial models (Fig. F2) to HELCOM's sub-divisions. For TN, TP and Chl a, specific targets are not given, but it is recommended to use the targets below the suggested values. Winter means are December-February and summer means are June-September.*

Basin	Winter		Summer	Annual				
	DIN	DIP	Chl *a*	DIN	DIP	TN	TP	Chl *a*
Kattegat	4.07	0.49	<1.22	1.52	0.21	<17.43	<0.64	<1.45
The Sound	3.34	0.42	<1.15	1.12	0.23	<17.33	<0.68	<1.12
Great Belt	5.00	0.59	<1.66	1.81	0.31	<20.95	<0.95	<1.67
Little Belt	7.09	0.71	<2.82	2.75	0.36	<23.29	<1.01	<2.26
Kiel Bay	5.45	0.60	<2.05	1.66	0.32	<22.20	<0.96	<1.88
Bay of Mecklenburg	4.24	0.50	<1.71	1.32	0.31	<21.65	<0.98	<1.70
Gdansk Basin	4.16	0.36	<2.19	3.04	0.20	<17.28	<0.55	<1.60
Arkona Sea	2.90	0.36	<1.83	1.21	0.22	<17.39	<0.67	<1.45
Bornholm Sea	2.52	0.32	<2.20	1.09	0.19	<16.05	<0.54	<1.00
Eastern Gotland Basin	2.59	0.29	<1.88	1.66	0.19	<16.51	<0.45	<1.13
Western Gotland Basin	1.97	0.33	<1.23	0.97	0.19	<15.08	<0.45	<0.25
Northern Baltic Proper	2.90	0.25	<1.76	1.17	0.14	<16.22	<0.38	<0.90
Gulf of Riga	6.72	0.41	<4.06	4.15	0.26	<37.46	<0.70	<4.99
Gulf of Finland	6.31	0.59	<4.03	3.03	0.25	<21.37	<0.55	<2.41
Åland Sea	2.67	0.21	<1.52	1.25	0.13	<15.60	<0.28	<1.04
Bothnian Sea	2.75	0.19	<1.52	1.57	0.13	<15.65	<0.24	<1.33
The Quark	3.68	0.10	<2.01	2.56	0.09	<17.29	<0.24	<1.55
Bothnian Bay	5.15	0.07	<1.61	4.27	0.07	<16.86	<0.18	<1.21

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